CHARACTERIZING NITROGEN LOSSES TO AIR AND DRAINAGE WATER FROM RED CLOVER MANAGED AS GREEN MANURE OR FORAGE

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ABSTRACT

The transfer of N from legume green manures (GMr) can satisfy the needs of a successive cash crop, but rotations that have over-wintering legumes also carry an increased risk of off-season (Sep.–June) N losses, especially during spring thaw. Spring-wheat yield among four GMr systems were evaluated with respect to off-season (GMr; Sep.–June) and in-season (wheat; June–Sep.) N_2O emissions, as well as full-year NO_3^- leaching and dissolved N_2O losses during spring-thaw from a tile-drained sandy loam soil in Atlantic Canada over 2 rotations (2011–2013). Four GMr systems (treatments) differed in the timing and season of GMr incorporation and the use of additional N as fertilizer or manure. The majority (66%) of cumulative N_2O emissions were measured during the off-season because of high N_2O emissions events during spring thaw. There was no clear effect of GMr system on these emissions, which may have been a result of the pattern and duration of soil freezing and thawing. Spring thaw also coincided with the highest dissolved N_2O concentrations (100–300 µg N_2O-N L^{-1}) in tile-drained water, which represented potential N_2O emissions of 21 to 116 g N_2O-N ha^{-1}. Belowground N_2O concentrations and soil water content measurements during winter provided further evidence of the relationship of N_2O dissolved in drainage water and N_2O emissions at the soil surface. Wheat yield among treatments in either year of study were not different, but was 1.5 times greater in Year 2 (2.62 ± 0.27 Mg ha^{-1}), than Year 1 (1.05 ± 0.12 Mg ha^{-1}). The highest NO_3^- concentrations in drainage water (Oct.; 13.8 mg NO_3^--N L^{-1}) were measured from the GMr system with the earliest fall incorporation (i.e., Sep.) and the addition of spring fertilizer when compared to the mean of all other treatments (9.8 mg NO_3^--N L^{-1}). The use of supplemental N did not translate into additional gains in yield, yet increased in-season N_2O emissions and greater NO_3^- leaching. Off-season N losses proved to be a substantial part of the annual N loss budget and dissolved N_2O in drainage water was identified as an additional pathway for N loss at spring thaw.

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LIST OF ABBREVIATIONS

BNF	Biological N_2 fixation
DM	Dry matter
DNRA	Dissimilatory nitrate reduction to ammonium
EF+N_{70}	Early Fall + N fertilizer at 70 kg ha^{-1}
GHG	Greenhouse gases
GMr	Green manures
HLF+M_{70}	Hayed Late Fall + liquid dairy manure; target N rate of 70 kg ha^{-1}
LF	Late Fall
N_{min}	Soil inorganic nitrogen (NH_4^+ + NO_3^-)
NUE	Nitrogen use efficiency
Q_{10}	Temperature coefficient
RSMN	Residual soil mineral nitrogen
SNS	Soil nitrogen supply
SP	Spring
WFPS	Water filled pore space
ΣFDH	Sum of Freezing Degree Hours (<0 ºC)
ΣTDH	Sum of Thawing Degree Hours (>0 ºC)
ΣDH	Sum of Degree Hours (ΣFDH+ ΣTDH)

1. GENERAL INTRODUCTION

1.1 Introduction

Although there is little evidence to support the claim that organically produced food has increased nutritional benefits (Magkos et al., 2003; Johansson et al., 2014), the North American consumer is more likely motivated by the "improved nutritional value" when purchasing organic food, while European consumers are focused on the benefits to "environmental stewardship" or animal welfare (Lund, 2006; Lynch, 2009). Including a diversified crop rotation and increasing soil inputs of crop residues and manures can benefit soil and plant health and cumulatively can reduce off-farm inputs of pesticides and soil nutrients. Research that takes more of a systems approach to the interacting and cumulative effects of the social, ecological, and economic goods and services (Lynch et al., 2014) could persuade the North American consumer of other benefits from the organic industry. Despite the limited organic agricultural research in Canada, the demand for organic food continues to grow, which contributes added pressure on organic farmers to increase yields while minimizing environmental degradation.

Organic farming systems in eastern Canada use biennial legumes as green manures (GMr) in forage–grain crop rotations to increase soil fertility and to improve overall soil conditions from the input of organic matter. With these improvements to soil quality, there is also an indication of increased N losses to air and drainage water when legumes are the dominant GMr plant species (Bergstrom and Kirchmann, 2004; Gregorich et al., 2005; Rochette and Janzen, 2005; Korsaeth, 2008; Jensen et al., 2012). Because biennial GMr accumulate large amounts of N in above and belowground biomass (Hatch et al., 2014), N losses are likely to occur outside of the main growing season before the planting of the intended crop—especially in eastern Canada where edaphic and climate conditions can encourage N loss.

The majority of the soil N cycle is mediated through the microbial community that begins when organic N is transformed to inorganic N, or atmospheric N_2 is symbiotically converted to a

biologically usable form (Robertson and Vitousek, 2009). Biological properties that are coupled to soil physical characteristics regulate O_2 availability and water movement through the soil profile (Blagodatsky and Smith, 2012), which can lead to gaseous and aqueous N losses to the environment. The greatest N loss pathway occurs as NO_3^- leaching; especially during periods of high precipitation and limited plant uptake when residual soil mineral nitrogen (RSMN) is high following harvest (Rasouli et al., 2014). Nitrous oxide also receives significant attention because 50 to 64% of total N_2O emissions are from agricultural soils (Janzen, 1998; Burton et al., 2008), as well as N_2O being associated with processes that lead to ozone destruction (Cicerone, 1989; Mosier et al., 1998; Butterbach-Bahl et al., 2013). The numerous N_2O production and consumption processes occurring in soil are likely occurring concurrently, but denitrification and nitrification activities are dominant in most arable soils (Firestone and Davidson, 1989; Pennock et al., 2006). Nitrification related processes are more dominant in semi-arid environments (e.g., Ma et al., 2008; Bedard-Haughn et al., 2013), while denitrification produces greater N_2O in more humid climates (Wagner-Riddle et al., 2008; Burton et al., 2012). There is also a risk of N_2O losses occurring from drainage tiles (Burton et al., 2012) because N_2O is highly soluble in water and has a high probability of downward movement in soil water when soil N_2O concentrations are elevated (Heincke and Kaupenjohann, 1999).

The timing and season of GMr incorporation may be modified in an effort to minimize N losses, but there are factors other than environmental N losses that are considered when cropping patterns are designed. In particular, the ability for a farmer to access a field with heavy machinery could be limited by high precipitation and saturated soil conditions. In an effort to regulate the timing of soil N availability, the season and timing of GMr incorporation can be modified, as well as the use of spring added supplemental N as fertilizer or manure. In eastern Canada, early fall incorporation has a high risk of off-season (i.e., fall to spring) N losses as soil mineralization and nitrification processes will likely not be limited by a lack of soil water or soil temperature. Spring GMr incorporation may allow for better protection against off-season N losses, but the soil N supply (SNS) may not be well synchronized with crop uptake and could potentially reduce cash crop yields. Delaying fall GMr incorporation until Nov. is an alternative cropping practice that has had some success in maintaining an adequate SNS while minimizing off-season N loss (Sanderson and MacLeod, 1994; Carter, 1994; Sanderson et al., 1999; Lahti and Kuikman, 2003). Decomposition is delayed at this time of year due to cooler soil

temperatures, but then proceeds the following spring without having to wait for ideal soil conditions for tillage, thereby reducing the amount of field preparation needed to prepare the seedbed of the succeeding crop. Removing aboveground GMr biomass as hay, or using it as forage with a later application of manure, are other GMr management options. Although GMr decomposition and N loss are highly dependent on annual weather patterns, several GMr management strategies could be used to minimize N loss while maximizing the transfer of N to the cash crop.

The likelihood of N losses due to leaching or gaseous emissions during the off-season is positively related to the soil NO_3^- content, but diminishes when a grass or legume crop is planted (i.e., catch crop) after harvest (Bouwman et al., 1993; Askegaard et al., 2011). The increase in the NO_3^- pool could either be a result of reduced uptake by the cash crop, or the increase could occur before cash crop planting due to early or rapid GMr decomposition. The focus around GMr tillage and the timing of incorporation is typically based on managing the aboveground biomass, yet much of the accumulated organic N from the fertility building phase of the rotation is located belowground in the rhizosphere (Walley et al., 2007; de Vries and Bardgett, 2012; Arcand et al., 2014). The contribution to soil inorganic N (N_{min}) from the belowground biomass has been estimated at 30–75% (Evans et al., 2003), but would be accelerated with tillage. Thus, all GMr management strategies, regardless of the seasons and timing of aboveground biomass incorporation, have the potential to increase soil NO_3^- availability and the risk of environmental N losses, especially without a growing crop during the off-season.

The overall objective of this study was to evaluate N losses from four GMr rotation systems as it moved from the GMr phase into the cash crop phase of the rotation. A secondary objective was to evaluate how the cash crop acted as an N sink as a result of the four different GMr systems. Having each phase of the rotation present in every year allowed us to examine two distinct years with respect to year to year weather differences. Treatments differed in management by the timing of GMr incorporation, and the use of supplemental N as fertilizer and dairy manure along with the removal of aboveground biomass. Nitrogen losses in this study included off-season soil N_2O emissions from the GMr phase of the rotation and in-season losses from a spring wheat crop. Total NO_3^--N load in drainage water was measured year-round and represented losses from the entire 4-yr crop rotation (Soybean–GMr–GMr–Wheat). In addition to

NO_3^- leaching, N_2O dissolved in drainage water was assessed through the winter and as soil thawed.

1.2 Organization of the Dissertation

Research presented in this dissertation is organized in manuscript format and presented in Chapters 3 through 5 following this introduction and the literature review in Chapter 2. All research chapters represent work conducted from the spring of 2011 until the spring of 2013 at one research site in Truro, NS, Canada. The site is located within an area that is certified for organic agricultural research and undergoes an annual audit of the agronomic practices used in accordance to Canadian standards for organic agriculture. The experimental site also includes a network of sub-surface drainage tiles where water quality and drainage volume can be evaluated from 12 isolated plots (16 x 75 m). The field experiment was designed as a four year crop rotation where each phase of the rotation was present every year—an aerial image of the site along with a schematic of the plot layout is found in Appendix 1.

The first research chapter (Chapter 3) describes N_2O and CO_2 emissions from a 2-yr-old red clover stand during the off-season (i.e., Sep.–June) over two years. Weather data along with soil water and temperature measurements are used as indices of physical properties occurring at the surface and within the soil profile that may have a control on the timing and magnitude of N_2O emissions. I hypothesized that GMr treatments that were incorporated in fall would have greater off-season N_2O emissions when compared to spring incorporated GMr, but delaying fall incorporation until Nov. would reduce these emissions. Because this study is solely focused on the GMr phase of the rotation during the off-season (Sep.–June) there is no use of supplemental N in any of the treatments.

The second research chapter (Chapter 4) investigates the below ground gas concentrations of N_2O and CO_2 within the 7.5 to 57.5 cm soil profile from GMr plots during the off-season. The content of N_2O dissolved in drainage water due to the four year crop rotation is also examined. Treatments and plots used in this study are identical to those used in Chapter 3. Belowground N_2O concentrations would act as an indicator for potential N_2O losses at the soil surface and dissolved N_2O leaving the site in drainage water was the tested hypothesis.

Chapter 5 outlines growing season (June–Sep.) soil N dynamics as a result of the varied GMr systems and plant N uptake and yield of spring wheat, while measuring in-season CO_2 and N_2O emissions and full-year NO_3^- leaching over the entire four year crop rotation. The hypothesis used in this study was that moderate additions of supplemental N, in the form of manure and mineral fertilizer, would improve yields of spring wheat as compared to using GMr alone without increasing N losses to air and drainage water.

A synthesis of the major findings from all three research chapters is discussed in Chapter 6 and evaluates the clover management systems with respect to the seasonal dynamics of N losses through the various pathways discussed above and their ability as a GMr to supply N to the successive crop.

2. LITERATURE REVIEW

2.1 Organic Agriculture and GMr

In contrast to natural ecosystems that have a relatively tight N cycle where peak availability of soil N coincides with maximal plant uptake, agricultural systems are notorious for low N efficiencies (Christensen, 2004). Decades of research have been dedicated to modern solutions that increase N efficiency by improved inorganic fertilizer formulations, improved plant cultivars, and the timing and method of soil inputs of organic N in the form of manures, composts, and plant residues. Although the principles are deeply rooted in the past, there has been a rise in organic agricultural research with the goal of designing a system that better mimics a natural system (Dawson et al., 2008). The ability to supply enough essential plant nutrients to satisfy crop yields without using inorganic fertilizers has not slowed the acceptance of organic agriculture, but designing a system that optimizes plant uptake with soil nutrient availability has proved to be one of the greatest challenges of organic agricultural systems (Berry et al., 2002; Watson et al., 2002).

Increasing N use efficiency (NUE)—defined here as the ratio of plant uptake to N input—in cropping systems is considered to be the best method to decrease environmental N losses (Davidson et al., 2014). In agricultural systems that rely on inorganic fertilizers the NUE is estimated to be around 50% (Conant et al., 2013), but has been increasing recently due to improved crop yields and N application rates that have leveled off in North America (Davidson et al., 2014). In organic systems, strategies to enhance NUE would include a better synchronization of the SNS from legume residues with varied timing of GMr incorporation and the use of supplemental N (Lynch et al., 2012), and use of deep rooted catch crops (Thorup-Kristensen, 2006). Other work being done that is specific to organic research, is focused on the breeding of plant varieties that are selected to be more productive in lower N environments because of more extensive root systems (Dawson et al., 2008) and is a promising strategy for the future. Regardless of the production practices used, reducing the amount of reactive N in the

environment is a critical goal for all agricultural systems (Galloway et al., 2002) so that yields continue to increase without further environmental pollution.

Although the yield gap between conventional and organic agricultural systems can be substantial (Dawson et al., 2008), benefits of organic agricultural practices on a soil's internal N cycle may require a longer time period before a pronounced effect on productivity becomes apparent. In comparison to continuous corn, a diversified crop rotation and greater organic matter inputs have increased yield of the cash crop in some long-term studies (i.e., Drury et al., 2014a); however, short-term changes to physical or chemical soil properties may be difficult to quantify following a shift in management (Nelson et al., 2009; Knight et al., 2010). Some of the key soil properties that benefit from increased inputs of organic matter include: improved soil structure (e.g., Tisdall and Oades, 1982; Thorup-Kristensen et al., 2003), increased soil water content and improvements to infiltration (Franzluebbers, 2002), greater nutrient availability from increased aggregate size and stability (Kong et al., 2007), and less soil degradation (Breland, 1995). Biological properties of soil quality may be more sensitive to changes in management, as compared to chemical or physical soil properties, especially when the distinct change in management comes from the increased input of organic matter. A larger microbial biomass has been associated with agricultural systems that include legumes in the rotation (McDaniel et al., 2014a), which has been shown to improve ecosystem N retention (de Vries and Bardgett, 2012) as well as the resistance to pests and disease in higher plants (Peoples et al., 1995). Crop rotations and the input of GMr residues can also increase net N mineralization (Sanchez et al., 2001), and the ability to decompose recalcitrant residues (McDaniel et al., 2014b). There are numerous benefits to soil properties from using a diversified crop rotation and increasing crop residue inputs, however, the possibility of negative consequences of greater N availability needs to be considered as N loss is positively related to N availability during periods without a growing crop.

The use of legumes grown as GMr is an essential component of soil fertility management plans on organic farms because the high costs or minimal availability of animal manures often restricts its use, leaving few other options for producers in these systems (Knight et al., 2010; Loes et al., 2011). Organic cropping systems in Atlantic Canada are characterized by extended (4–5 yr) rotations that typically include two years of leguminous GMr that are grown as forage, or to build soil fertility (Lynch et al., 2008; Lynch, 2009; Nelson et al., 2009). Red clover

(*Trifolium pratense* L.) is a biennial/perennial species that can be grown as a winter cover crop (Gentry et al., 2013), or maintained for multiple years when biomass is either removed and used as forage (Thilakarathna et al., 2012), plowed under in spring or fall (Thorup-Kristensen and Dresboll, 2010), or cut and left on the surface as a mulch (Hatch et al., 2014; Woodley et al., 2014). The amount of N supplied to a succeeding crop will vary with the species grown, edaphic properties, and the timing and method of termination (Liebman et al., 2012). However, for every Mg of cereal grain produced, the soil needs to supply 20 to 40 kg N ha^{-1} to satisfy adequate seed and vegetative production in these crops (Peoples et al., 1995). Estimates of biological N_2 fixation (BNF) by red clover fluctuate widely (Watson et al., 2002), yet on average 25 kg N Mg^{-1} of biomass is fixed from the atmosphere in aboveground portions of the plant (Peoples and Baldock, 2001; Carlsson and Huss-Danell, 2003), while 30 to 40 kg N has been estimated when belowground portions of the plant are included (Peoples et al., 2009). Therefore, for systems relying on GMr for N fertility alone, one or two Mg dry matter (DM) ha^{-1} of above and belowground GMr biomass is needed for every Mg of grain produced. This estimate works on the assumptions that N inputs from the native soil organic matter and N losses somewhat cancel each other out, and legume BNF is maximized. However, like conventional producers who apply fertilizer in excess of crop demand as an inexpensive insurance policy (Davidson et al., 2014), organic producers need to maximize yields by adding N in excess of crop demand, but the timing of the soil N supply (SNS) can be much more difficult to predict. Determining best management of GMr in organic production systems that produce acceptable yields with minimal N losses require further study (Liu et al., 2010), especially for humid maritime climates in eastern Canada.

2.2 Soil Nitrogen Transformations

Optimum conditions for biological activity and N mineralization occur around 30°C and a soil water content that is 50 to 80% of field capacity, or a water filled pore space (WFPS) of 60% (Bouwman et al., 1993; Voroney, 2007; Whalen and Sampedro, 2010). Soil biological activity can be compared among systems by the increase in production of CO_2 with temperature; commonly described by a Q_{10} temperature function of around two, over a 15 to 35°C soil temperature range (Voroney, 2007)—Q_{10} is the proportional change in respiration with a temperature increase of 10°C. Although activity in colder soils is much slower, the Q_{10} value for

soil temperatures close to 0°C were recently reviewed to be 4.8 over a range of cold soil environments (Hamdi et al., 2013). In these conditions, N transformations are still occurring, although different microbial species with unique temperature sensitivities are involved.

Nitrous oxide is of particular interest as a GHG because of its global warming potential, which is 298 times that of CO_2 (molecule for molecule) (Mosier et al., 1998), and its role in the destruction of stratospheric ozone (Cicerone, 1989; Butterbach-Bahl et al., 2013). Nitrous oxide is mainly produced through aerobic and anaerobic denitrification pathways, and as a by-product of nitrification soil processes (Firestone and Davidson, 1989; Davidson et al., 1991). The chemical decomposition of hydroxylamine during autotrophic and heterotrophic nitrification, and chemodenitrification of soil nitrite are likely only small contributions to overall N_2O production (Butterbach-Bahl et al., 2013). Aerobic processes include nitrifier-denitrification from organisms involved in nitrification as well as coupled nitrification–denitrification by distinct organisms of each process when soils transition to anaerobic status (Wrage et al., 2001; Kool et al., 2011). When soil O_2 becomes limited, denitrification can occur from a diverse group of organisms that use N oxides as alternative electron acceptors during respiration (Burton et al., 2012). Nitrate ammonification, or dissimilatory nitrate reduction to ammonium (DNRA), is yet another process that produces N_2O in soil (Schmidt et al., 2011).

Availabilities of soil NO_3^- and C, in addition to aeration, are known to be important factors controlling denitrification (Firestone and Davidson, 1989; Davidson, 1991); however, the ratio of NO_3^- to C and soil temperature have a strong control on N_2O:N_2+N_2O molar ratio during this process (Beauchamp et al., 1989; Morkved et al., 2006; Gillam et al., 2008; Mitchell et al., 2013). Because of the greater energy potential of NO_3^- over N_2O, NO_3^- is preferentially used as a terminal electron acceptor over N_2O by denitrifying organisms (Firestone et al., 1979; Cho et al., 1997), which can slow the complete reduction of N_2O to N_2. The availability of C influences N_2O production because of its role as an energy source and electron donor for heterotrophic organisms involved in N_2O production. For example, at low NO_3^- levels and increased C availability, the N_2O:N_2+N_2O molar ratio is decreased; resulting in a greater proportion of N_2 being produced during denitrification (Kramer et al., 2006; Mitchell et al., 2013), whereas N_2O emissions are favored as NO_3^- and C availability increase (Miller et al., 2008; 2009). Furthermore, increased microbial activity when C availability is high can also decrease soil O_2 availability, creating conditions conducive to denitrification (St. Luce et al., 2011; Burton et al.,

2012). Indeed, the relative composition of NO_3^-, labile C, and O_2 in a soil has a complex control on N_2O production and emissions from soil.

Emissions of N_2O have been positively correlated with soil inorganic N intensities, calculated as the time-weighted sum of NO_2^- + NO_3^- (e.g. Burton et al., 2008: Zebarth et al., 2008; Asgedom et al., 2014) or NO_2^- over a defined season (e.g., Maharjan and Venterea, 2013). Inorganic N intensity is then used as an index of the magnitude and duration of inorganic N exposure to microbes involved in N_2O production. Nitrite intensity explained a greater share of the variability in N_2O emissions over NH_4^+ or NO_3^- in a fertilizer source and placement study in Minnesota, USA (Maharjan and Venterea, 2013), as well as a ^{15}N tracing study that observed NO_2^- being the typical precursor to N_2O production in the laboratory (Russow et al., 2009). The lack of routine soil NO_2^- measurements because of added cost and the necessity of immediate analysis explains why NO_2^- as a substrate for N_2O production is not traditionally expected or discussed discussion with N_2O production in most agricultural studies. However, NO_2^- rapidly reacts to form N_2O under a broad range of aerobic conditions (Venterea, 2007) and has a closer proximal position to N_2O in most N_2O producing mechanisms when compared to NO_3^- (Venterea and Rolston, 2000) and should be considered when evaluating management practices that reduce soil accumulation of NO_2^- may also reduce N_2O emissions (Maharjan and Venterea, 2013).

2.3 Timing of GMr Incorporation

Net mineralization and nitrification in soil at a particular point in time is best described as an index of the competing processes of immobilization and mineralization (Schimel and Bennett, 2004), and ultimately plant uptake and N losses (St. Luce et al., 2011). Mineralization will be dominant over immobilization when added crop residues have a C:N ratio that is less than 20 (Thorup-Kristensen et al., 2003) and will also affect the overall rates of these processes occurring over time. A comparison of crop rotations with and without legumes found that a smaller overall amount of biomass returned to the soil from the rotation with legumes produced similar gross mineralization and nitrification rates over a growing season in a semi-arid environment because of the greater quality (i.e., C:N) of the legume residue (Bedard-Haughn et al., 2013). Crop sequence within a rotation can have significant effects on mineralization rates from year to year,

as available C and N pools are influenced by the preceding crop and will vary with management (Doran et al., 1987), yet these rates are difficult to predict across varying soil conditions and year to year (Phillips, 2008).

In temperate climates, fall GMr incorporation may increase over-winter NO_3^- leaching as N immobilization processes are more sensitive to low soil temperatures than nitrification (Hoyle et al., 2006), and mineralization (Andersen and Jensen, 2001; Magid et al., 2001; Lahti and Kuikman, 2003), although results may differ in the presence of a readily available C source (Cookson et al., 2002). Alternatively, spring incorporated GMr eliminates winter fallow and the risk of increased N loss, but establishment of the succeeding crop may be hampered by such factors as prolonged N immobilization (Francis et al., 1992), reduced germination (Lynch, 1980), and increased competition from annual and perennial weeds due to the lack of weed control during the GMr phase of a rotation (Askegaard et al., 2011). Finally, fall GMr incorporation may be more desirable, as spring field preparation may be delayed in heavier soils during years with heavy precipitation or excessive snowmelt because of saturated soil conditions that limit field accessibility with heavy equipment. Soil properties and climatic conditions will influence the timing of GMr incorporation. Having a variety of GMr incorporation options to ensure minimal N losses and a sufficient soil N supply to satisfy cash crop needs is necessary.

Fall and spring incorporation of GMr for spring wheat production in Finland was examined by Lahti and Kuikman (2003), both in the field and laboratory, where treatments included early fall incorporation (Sep. 1), delayed incorporation (Oct. 20), late fall (Dec. 23), and spring (May 3). Greatest wheat grain yields were achieved from late fall incorporation (4.03 Mg ha^{-1}) as compared to early incorporation in September (lowest yield: 2.79 Mg ha^{-1}). This contradicts the findings of Francis et al. (1995) who did not observe a treatment effect on wheat yield from delayed fall GMr incorporation. However, there was a yield depression when GMr was incorporated in spring attributed to immobilized N early in the growing season (Francis et al., 1995). Understanding how the timing of GMr incorporation influences yields of the succeeding crop, and the associated pathways for N loss, will provide organic producers with options that can be designed for their soil conditions and production systems.

2.4 NO_3^- Leaching

Nitrogen is lost from the root zone as a result of NO_3^- leaching, which is thereby controlled by N mineralization and nitrification rates, crop uptake, manure or fertilizer additions, precipitation, soil thawing, and the timing of tillage (Dinnes et al., 2002). The loss of inorganic N occurs predominately as NO_3^- because NH_4^+ is held much more tightly in the soil matrix because of its positive charge and the predominance of negative charged exchange sites in soil (Brady and Weil, 2001). An increasing number of agricultural soils in eastern Canada have been classified as having a high risk for NO_3^- leaching, especially between growing seasons (Milburn et al., 1997; De Jong et al., 2007; Drury et al., 2007; Lynch, 2009). In areas of potato cultivation on PEI, NO_3^- concentrations in soil leachates can reach as much as 65 mg L^{-1} (Savard et al., 2007), but diminish through the winter (Zebarth et al., 2003). Synchronizing the SNS with crop uptake and the inclusion of cover crops in GMr systems can reduce NO_3^- leaching (Francis et al., 1992; Drinkwater et al., 1998; Baggs et al., 2000; Stopes et al., 2002; Lahti and Kuikman, 2003; Eriksen et al., 2004; Jabloun et al., 2015). However, microbial mediated N transformations do not solely occur during the period of crop uptake, and could be significant when the timing of GMr tillage occurs outside of the main growing season (Dahlin et al., 2005), especially in fields with sub-surface drainage tiles (Drury et al., 2014b).

The use of animal manures along with a range of organic farming practices were investigated for N leaching during 3 full rotations of a 4-yr crop rotation in Denmark (Askegaard et al., 2011), where the application of conventional animal manure is limited to 70 kg total N ha^{-1} yr^{-1} in an effort to limit excessive NO_3^- leaching. A three treatment factorial experiment included the following factors: (1) proportion of grass/clover in rotation, (2) with and without an over-winter catch crop, and (3) with and without animal manure. There was no effect of manure application on N leaching, which implies that moderate manure applications in the spring will likely not lead to reduced NUE or a large residual N pool. The biggest contributor to N leaching was the inclusion of an over-winter cover crop, and more specifically, the frequency and intensity of fall tillage.

Soil inorganic N (N_{min}) content from the 0–30 cm soil depth on 228 commercial potato fields over three years was sampled at planting and harvest to relate N content to cropping history and weather in New Brunswick, Canada (Zebarth et al., 2003). Although there was a high

degree of variability among fields and years, low NO_3^- contents in spring were related to years that had precipitation amounts greater than normal in the preceding fall and spring and higher NO_3^- contents in spring were related to relatively drier conditions. Soil N_{min} in spring was controlled by climatic conditions and the history of the preceding crop. More specifically, low NO_3^- contents at the 0–30 cm soil depth were likely a result of leaching and/or denitrification in wet years while high soil N_{min} in spring (e.g., 2 to 124 kg N ha^{-1}) was as a result of a preceding red clover or hay crop (Bélanger et al., 2000; Zebarth et al., 2003).

2.5 Legume GMr and N_2O Emissions

Soil NO_3^- levels not only influence N loss through leaching, but are positively correlated with N_2O emissions as well (Bouwman et al., 1993; Chirinda et al., 2010a; 2010b). Emissions of N_2O from Canadian agricultural operations are estimated to be 80 Mg N_2O-N yr^{-1}, or 60% of the total anthropogenic related emissions (Janzen, 1998) and are primarily associated with denitrification soil processes in humid soil moisture regimes (Mosier et al., 1998; Burton et al., 2008). During the GMr phase of a rotation, N_2O production is likely because of increasing off-season NO_3^- due to favorable soil conditions in the fall as a result of higher water contents (i.e., decrease in air-filled soil porosity), and a flush of available C as plant roots and high quality (<20 C:N) crop residues start to decompose. A fall tillage experiment conducted on a light textured soil in Denmark reported N_2O emissions that were 69% greater in the fall following conventional tillage and the incorporation of crop residues as compared to reduced and no-till systems (Mutegi et al., 2010). Best management plans for crop rotations that include legume GMr need not only consider N availabilities to the successive crop, but the potential for over-winter N loss as well.

When GMr residues are soil incorporated, net mineralization is the likely outcome over immobilization because of the high quality (i.e., <20 C:N) organic matter. Soil N mineralization rates and N_2O flux can increase with the incorporation of GMr residue (Aulakh et al., 1983), but are also shown to reduce cumulative growing season emissions when compared to cropping practices using fertilizers (Osterholz et al., 2014). Furthermore, the flush of microbial activity can deplete soil O_2 concentrations that may promote denitrification. Chirinda et al. (2010a) evaluated emissions of N_2O from three organic arable crop rotations and one conventional system in Denmark with the hypotheses that organic rotations would have lower N_2O emissions

because of restricted N availability and that spring emissions would increase following the incorporation of catch crops and grass-clover leys. Both hypotheses were rejected in their study, as cumulative emissions were not different among cropping systems and spring incorporation of plant residues failed to provide any clear pattern in N_2O emissions.

Although organic systems do not use mineral fertilizers, which are typically associated with increasing N_2O emissions, the transformation of GMr residue into inorganic N can stimulate N_2O production as well. Ellert and Janzen (2008) measured higher cumulative (3-yr) N_2O emissions in a 5-yr crop rotation when wheat and followed three years of continuous alfalfa as compared to a corn-wheat-corn sequence. The use of GMr in rotation may also stimulate N_2O emissions that occur during spring thaw, especially when incorporated in the fall. Following Sep. alfalfa plowdown in Ontario, Wagner-Riddle et al. (2007) measured N_2O emissions for Oct. at 1 kg N ha^{-1} mo^{-1}, but emissions later reached a minimum in Dec, and then increased again at spring thaw (ca. 1.2 kg N ha^{-1} mo^{-1}). However, no-till and perennial legume GMr could also stimulate N_2O emissions because of the effect that tillage has on soil pore size and continuity that modifies soil water storage and movement, such as water filled pore space (WFPS) and air entry. Ball et al. (2008) found elevated N_2O emissions were 1.5 to 35 times higher from no-till treatments as compared to bare soil surfaces that had received inversion tillage. The greater WFPS and greater continuity of soil pores through the profile were said to have a strong control on N_2O production and emissions, which suggests that emissions were not solely connected to the timing of residue incorporation, but the temporal regulation of water and air in soil pores has a significant control on N_2O production and emissions. However, there are contrasting results when N_2O emissions from no-till studies are examined with greater emissions being measured in poorly aerated soils under no-till (Rochette, 2008), and a reduction of N_2O emissions when no-till is practiced for more than ten years and fertilizer N is placed below the 5-cm soil depth (van Kessel et al., 2013).

2.6 Off-Season N_2O Emissions

Studies that look at different management strategies of GMr incorporation and the associated N loss from sub-zero soil temperatures are needed (Phillips, 2007), as many of the unknown N losses from cropping systems that rely on organic N sources are suspected to be

from denitrification losses during the dormant season (Phillips, 2008). Because of the reduced availability of C and N substrates or O_2 diffusion to microbial populations (Christensen and Tiedje, 1990; Phillips, 2007; Clark et al., 2009), soil processes responsible for N transformations in arable soils are assumed minimal in sub-zero soil temperatures (Larionova et al., 2007). Risk et al. (2013) reviewed the mechanisms involved in elevated N_2O emissions during winter and spring thaw as being a combination of: (1) a release of N_2O produced under a basal soil ice layer that later diffuses out of the soil profile as topsoil thaws (Burton and Beauchamp, 1994; van Bochove et al., 2001), and the off-gassing of N_2O dissolved in soil water with an increase in temperature (Goodroad and Keeney, 1984; Heincke and Kaupenjohann, 1999), (2) an increase in substrate availability that stimulates the microbial population (*de novo*) (Christensen and Tiedje, 1990; Koponen et al., 2006), as a result of aggregate turnover and the subsequent release of labile organic matter (Bullock et al., 1988; Christensen and Tiedje, 1990; Six et al., 2000; Phillips, 2008), and (3) the activity of specific enzymes and gene expression associated with microrganisms involved in N_2O production and consumption processes (Wertz et al., 2013; Németh et al., 2014; Tatti et al., 2014).

Considerable soil N_2O flux has been recorded during spring thaw (Rover et al., 1998; van Bochove et al., 2000b; Jacinthe et al., 2002; Wagner-Riddle et al., 2007; Phillips, 2007; 2008; Wagner-Riddle et al., 2010). Spring thaw accounted for up to 70% of the annual N_2O emissions from soils (Wagner-Riddle et al., 2008) and were comparable or higher than growing season fluxes from snow-covered soils during winter and spring (van Bochove et al. 2000a). A bi-modal distribution of N_2O emissions at spring thaw has been described as being an initial physical release of N_2O (Risk et al., 2014), followed by more rapid and vigorous emissions events newly produced at the soil surface as temperatures fluctuated between the freezing point and 5°C (Wagner-Riddle et al., 2010). Risk et al. (2014) determined that around 75% of the spring thaw N_2O emission was due to an increase in biological activity at the soil surface (Wagner-Riddle et al., 2010), as compared to emissions that resulted from passive degassing, as described by van Bochove et al. (2001). The extent (i.e., depth, duration, temperature) of soil freezing has been positively related to the size of spring thaw N_2O emission (Christensen and Tiedje, 1990; Wagner-Riddle et al., 2007; Glenn et al., 2012; Vankoughnett and Henry, 2013); an effect that become especially relevant when a change in climate predicts lower snowfall and more winter

rain events (van Bochove et al., 2001; Henry, 2008), which could lead to a lower snowpack and soil temperatures that would vary more with air temperature.

2.7 Soil Profile N_2O Concentrations

The composition of soil air at a given point in time is a result of the rate of production and consumption of gases, diffusion of gases through the soil pore spaces, and the partitioning of the gas within the liquid, gaseous, and solid phases of the soil (Farrell et al., 2002). The change in concentration with depth has commonly been used to predict surface flux using the gradient method, as reviewed by Maier and Schack-Kirchner (2014). However, the prediction of effective diffusivity is a major source of uncertainty in these calculations, which are typically derived from air-filled pore-space measurements, without the consideration of how ice at the soil surface affects diffusion or drainage properties. Therefore, in soils that experience freeze–thaw cycles on a daily cycle, the use of this technique is not suitable for predicted surface flux. However, the changes in concentration during freeze–thaw cycles does provide insight into any restrictions to gas diffusion and the potential for N_2O production and can then be related back to field management or weather patterns. Additionally, Yanai et al. (2011) measured O_2 concentrations at 10 and 30 cm soil depths in conjunction with N_2O concentrations at similar depths during a thawing event and determined that N_2O concentrations at depth decreased following a surface N_2O emission event at same time as the soil O_2 concentration increased to ambient atmospheric levels. A frozen soil layer that restricted soil gas diffusion to the atmosphere and limited O_2 availability in the soil profile was thought to have caused these results.

During freeze–thaw related events, N_2O is emitted from the soil surface due to changes in solubility with temperature, production near the soil surface, and diffusion from depth (Goodroad and Keeney, 1984; Burton et al., 1997; Risk et al., 2013). Risk et al. (2014) described N_2O emissions at spring thaw as two distinct events from a silt loam soil near Elora, ON. The first N_2O emission event was from 12 to 21 Mar. when a total of 3.55 mg N_2O m^{-2} (35.5 g ha^{-1}) was measured with a simultaneous N_2O mass decrease of 14.1 mg N_2O m^{-2} from the soil profile (0–70 cm). The second N_2O emission event occurred from 21 Mar. to 3 Apr. with a cumulative emission of 12.91 mg N_2O m^{-2}, yet the change in N_2O mass within the soil profile was only 2.49 mg N_2O m^{-2}. In total, 25% of the accumulated N_2O within the soil profile was assumed to have

contributed to a surface flux of N_2O during the first 10 days of spring thaw. A larger N_2O surface emission event (12.91 mg N_2O m^{-2}) over the next two weeks was then attributed to *de novo* production closer to the soil surface. In a variety of landscape positions in Manitoba, N_2O accumulated within the soil profile before spring thaw, but profile N_2O from the lowest position (riparian) were assumed to have been further reduced, as no considerable N_2O emissions from the surface were recorded (Gao et al., 2014). Although N_2O has been shown to accumulate within the soil profile over winter, a significant flux at the soil surface during thaw periods does not always occur. These contrasting results highlight the dynamic nature of N_2O production and release from the soil surface during spring thaw.

2.8 Hydrological Soil Properties and N_2O Emissions

The potential for nitrification and denitrification as related to WFPS were first described by Linn and Doran (1984). Although measures of WFPS give an approximate idea of the aerobic status of a soil, they do not describe the size distribution of soil pores and thus, the overall ability to drain water and accept air at the soil surface. Balaine et al. (2013) described a positive relationship between maximum N_2O emissions and soils air-entry potential (r^2 = 0.96), which implied that high N_2O emissions are not just associated with high denitrification potential, but that emissions occur when soil pores are not completely blocked with water. Similarly, three landscape positions (e.g., ditch, near-ditch, mid-field) that varied in bulk density (0.7–1.2 g cm^{-3}) and total C (11.1–15.6 g kg^{-1}) from a poorly drained, silty soil in Maryland had different levels of water content, but a similar range of matric potential (–3.75 kPa) at the time of maximum N_2O emissions from intact soil columns (Castellano et al., 2010). A matric potential of –3.75 kPa represents a pore radius of 39.58 µm, which indicates that maximum N_2O flux occurred as pores of this size have completely drained of water. Comparing field operations from different areas and climates then becomes challenging, without a clear understanding of how the hydrological properties of a soil compare. Greater daily N_2O emissions (440 to 740 g N ha^{-1} d^{-1}) following precipitation events and N fertilization were measured from fine textured soils in southern Manitoba (Glenn et al., 2012) when compared to a field (150 g N ha^{-1} d^{-1}; Wagner-Riddle et al., 2007) that had actually received greater N fertilization, but had better drainage and less available C. Similarly, sharp peaks in N_2O emissions under wet no-till conditions were produced by

greater water contents as compared to tilled soil, without a further reduction to N_2 (Ball et al., 2008). Finally, two soils with similar total porosity, but one with greater macro-porosity (i.e., 3000 μm) than the other also had different N_2O emission patterns (van der Weerden et al., 2012). Larger pores with greater connectivity provided faster soil water infiltration and lower cumulative emissions, which allowed more air to occupy soil pores, and thus, a lower potential for denitrification and N_2O emission. Models that predict the pore size and continuity in soils are expected to become increasingly important to our understanding of the mechanisms that control gas production, transport, and release to the atmosphere (Ball, 2013). Soil water content and the size and distribution of soil pores have a strong control on the aeration status of a soil, which is ultimately controlled by incoming precipitation and a soil's ability to infiltrate water.

The pattern and extent of soil freezing will be affected by tillage, because of the change in soil pores and a modification of surface conditions, such as vegetation, that will trap more snow and insulate the soil from freezing and thawing cycles (Ling and Zhang, 2004). During soil freezing, smaller pores will freeze last because of osmotic and matric potentials that keep water from freezing (Harrysson Drotz et al., 2009). In contrast, smaller pores will thaw before larger pores because of the smaller volume of water and amount of latent heat needed to complete the phase change (Kane et al., 2001). The amount of drainage becomes a good indicator of the soil's ability to transmit water and gas at the soil surface because water acts as a gas diffusion barrier from the soil to the atmosphere when all soil pores at the soil surface are filled with ice (Cherkauer and Lettenmaier, 1999; Ireson et al., 2013; Watanabe et al., 2013). When soil experiences freeze–thaw dynamics, soil pore characteristics and hydrological properties might have an even greater control on N_2O production and related emissions. The timing of tillage in the fall will also have an effect on the size and continuity of soil pores during the off-season (Parkin et al., 2013), which will affect not only the rate and extent of soil freezing, but the rate and timing of soil thawing and the subsequent rapid infiltration of water from the surface and through the profile.

2.9 Dissolved N_2O

The first discussions about the amount of dissolved N_2O that exits a field in drainage tiles were initiated by Dowdell et al. (1979) who estimated that 0.25 kg N ha^{-1} could be lost from 250

mm of precipitation (Nov.–Mar.) in the U.K. During their study period, the amount of N_2O lost in drainage tiles equaled surface emissions measured at the site; they concluded that during periods of high N_2O concentrations and rapid drainage, dissolved N_2O losses are significant. Since then, various other studies have measured or estimated relatively low concentrations in drainage water and have therefore determined this to be an insignificant pathway for N loss for most agricultural areas. Nitrous oxide is highly soluble in cold water (Heincke and Kaupenjohann, 1999) and becomes a likely pathway for N losses when arable soils are equipped with sub-surface drainage systems that can contribute to global agriculture related GHG emissions (Baulch et al., 2011). Roper (2009) measured differences in dissolved N_2O collected from drainage tiles from a sandy soil under continuous and no-till treatments. No difference in tillage treatment (till vs. no-till) was observed over a 1.5 yr sampling period, yet there was a similar seasonal variation in dissolved N_2O (μg N_2O L^{-1}) in drainage water from both treatments that was influenced by a similar pattern of freeze–thaw events. The concentration of NO_3^- in collected drainage water during the same sampling times did not show the same variation, but cumulative loads of NO_3^- and N_2O were significantly correlated. The difference in NO_3^- and N_2O concentrations highlighted the contrasting soil environment that is experienced in soils that experience freeze thaw–events, but the positive relationship between cumulative losses of each suggests that the volume and pattern of drainage events had a control on overall N losses. The contribution of accumulated soil gas to surface emissions and the movement in drainage water is controlled by physical soil properties such as air-entry values, and gradients of water potential and temperature (Smith et al., 2003; Blagodatsky and Smith, 2012).

3. OFF-SEASON SOIL N_2O EMISSIONS AS AFFECTED BY THE TIMING OF GREEN MANURE TILLAGE

3.1 Preface

Synchronizing soil N availability with plant uptake is the goal for systems that use green manures (GMr) for soil fertility because an improvement to N efficiency also minimizes environmental N losses to air (N_2O) and drainage water (NO_3^-). Biennial GMr crops generate considerable amounts of N in above- and below-ground biomass that is maintained through the off-season, prior to cash crop planting. Spring incorporation is thought to have the lowest risk for off-season N losses, but the ideal soil conditions needed for inversion tillage may not align with all of the other field activities occurring in the spring. As a result, producers may incorporate the GMr crop early or late into the fall when soil conditions and time are more suitable. A considerable proportion of annual N_2O emissions have been measured during spring thaw; thus, it is of particular interest to understand how fall or spring GMr management affects off-season N_2O emissions. The length, depth, and frequency of soil freezing and thawing has been suggested to be a good predictor of N_2O emissions, while in other studies there is no clear edaphic property that predicts N_2O emissions at spring thaw. The goal of this study is to measure N_2O emissions from a red clover GMr crop through two consecutive off-seasons with respect to the timing of GMr incorporation and the removal of aboveground biomass. Because N_2O emissions measured during spring thaw can be significant, there is a focus on understanding the impact of soil freeze/thaw dynamics on these emission events.

3.2 Abstract

Off-season N_2O emissions can be a major proportion of annual emissions, but little is known about the timing of these emissions from leguminous green manures (GMr), especially during spring thaw. Over two years, we measured off-season (Sep.–June) N_2O emissions from a biennial GMr–spring wheat rotation in Atlantic Canada where GMr was incorporated in Sep. (Early Fall; EF), Nov. (Late Fall; LF), and May (Spring; SP). An additional treatment removed GMr biomass as summer hay before Nov. incorporation (Hayed Late Fall; HLF). Off-season emissions ranged from 0.41 to 0.95 kg N_2O-N ha^{-1} in Year 1 and 0.26 to 1.2 kg N_2O-N ha^{-1} in Year 2 with the largest daily emissions of 45.34 and 59.82 g N_2O-N ha^{-1} d^{-1} measured from SP and the mean of HLF and LF during spring thaw, respectively. All three fall incorporated clover treatments led to similar cumulative N_2O emissions during the off-season, but EF had 45 to 78% greater fall (Sep.–Jan.) emissions than all other treatments in Year 1 and 2, respectively. Delaying GMr incorporation until May (Spring; SP) produced different cumulative (Sep.–June) N_2O emissions when compared to all fall treatments in both years, however these differences were in different directions. In Year 1, SP was nearly twice as large as compared to fall treatments (950 vs. 494 g N_2O-N ha^{-1}), while fall treatments were 4.3 times as great as SP (262 g N_2O-N ha^{-1}) in Year 2. Treatments with colder and deeper soil frost (i.e., LF, HLF) had the highest emissions at spring thaw in Year 2, while the interaction of soil warming and the placement of GMr residue may have had a stronger influence in Year 1. However, there was a consistent relationship between years when peak N_2O emissions at spring thaw occurred during the rapid increase in soil temperature and water drainage. The season and timing of GMr incorporation may modify these properties that control the timing of emissions, which suggests that hydrological properties through the entire soil profile have a significant control on N_2O emissions during spring thaw.

3.3 Introduction

Nitrous oxide (N_2O) is a persistent greenhouse gas (GHG) that had an atmospheric concentration of 324.2 ppb in 2011, which was an increase of 5-ppb since 2005 (Hartmann et al., 2013). Although the contribution of N_2O to overall radiative forcing is small (e.g., 6.0–6.24%; Butterbach-Bahl et al., 2013; Myhre et al., 2013), the fact that N_2O is the primary ozone

depleting substance in the atmosphere (Cicerone, 1989; Ravishankara et al., 2009; Wuebbles, 2009) further emphasizes the need to understand how management decisions affect N_2O emissions. The production of N_2O as a result of agricultural activities contributes roughly 30 to 65% of atmospheric concentrations (Butterbach-Bahl et al., 2004; Ambus et al., 2011) and these emissions are primarily produced through denitrification and nitrification soil processes (Butterbach-Bahl et al., 2004; Pennock et al., 2006). Emissions of N_2O are typically associated with the rate and timing of N inputs, soil tillage, increased soil water, and freeze–thaw events (Burton et al., 1997; Gregorich et al., 2008; Risk et al., 2013), yet the proportion of N_2O produced during denitrification as compared to N_2 is a function of C and NO_3^- availability, in addition to the availability of oxygen in the soil (Butterbach-Bahl and Dannenmann, 2011).

Organic agriculture relies heavily on the use of animal manures and the use of legume species as cover crops or GMr, yet the contribution of N_2O from legume residues is not well understood (Rochette and Janzen, 2005). Increased N_2O emissions in GMr systems are due to the rapid release of inorganic N (N_{min}) from high quality (i.e., C:N < 20) legume residues (Baggs et al., 2000; Huang et al., 2004; Rochette and Janzen, 2005; Mitchell et al., 2013). This has led to a greater interest in determining best management plans for potato and other high-value crops in cool maritime climates (Sharifi et al., 2009; Lynch et al., 2012; Snowdon et al., 2013; Wertz et al., 2013), which could include managing the timing and season of GMr incorporation. Fall incorporation of GMr would allow for an earlier spring planting date in areas with a shorter growing season (Carter, 1992), but the absence of a growing crop following fall GMr plowdown poses a greater risk for N_2O emissions (Phillips, 2008; Ball et al., 2014). Spring GMr incorporation is commonly adopted with the intent to minimize N losses (Baggs et al., 2000; Lynch et al 2012; Ball et al., 2014). However, spring incorporation can delay N uptake of the succeeding crop due to immobilization or slow microbial N transformations from excess soil water and cool soil temperatures in some areas (Aulakh et al., 1991; Cookson et al., 2002; Lahti and Kuikman, 2003; Thorup-Kristensen et al., 2003; Woodley et al., 2014). Alternatively, N mineralization may occur well in advance of cash crop planting (Chapter 5), which may lead to increased N losses and diminished yields.

In addition to fall or spring GMr incorporation, other management options include delaying fall incorporation until soil temperatures are cool enough to slow soil mineralization and nitrification processes, as described for red clover–potato rotations (Sanderson and

MacLeod, 1994; Sanderson et al., 1999) and spring cereals in Atlantic Canada (Carter, 1994). Nitrification was inhibited in soil incubated at 2 or 5°C due to the slow acclimation of nitrifying organisms in cold soils (Cookson et al., 2002), which would reduce the risk of N losses from the reduction in NO_3^- intensity (Burton et al., 2008). The ability to limit nitrification through delayed GMr incorporation in the fall is a promising management option that could reduce N_2O emissions through the off-season. An alternative cropping strategy could involve cutting or grazing aboveground biomass as forage prior to fall incorporation, with the possibility for supplemental manure additions in the spring (Brozyna et al., 2013; Froseth et al., 2014).

The carryover of large amounts of organic C and N in GMr biomass through the winter is of particular concern for off-season N_2O emissions, especially at spring thaw, when N_2O emissions have been reported to be a considerable proportion of annual emissions (Duxbury et al., 1982; Rover et al., 1998; Wagner-Riddle et al., 2007; Matzner and Borken, 2008; Johnson et al., 2010; Yanai et al., 2011; Glenn et al., 2012). Both freezing and thawing intensity at the soil surface, measured as accumulated degree hours (ΣDH), have been suggested as controls over spring thaw N_2O emissions (Christensen and Tiedje, 1990; Wagner-Riddle et al., 2007; Glenn et al., 2012; Maas et al., 2013). Furthermore, Brumme et al. (1999) suggested that the duration of event-based emissions, such as those at spring thaw, are controlled by the soil environment, which is modified by soil re-wetting and freeze–thaw events, whereas the pattern of background emissions through the year relates to site-specific conditions, such as surface characteristics (e.g., bare or vegetated soil), and drainage properties. The interaction of soil physical and biological processes responsible for N_2O production are intimately connected by water filled pore space and soil diffusivity, which have been identified as good predictors of N_2O emissions (Ball et al., 2008; Rees et al., 2013; Petersen et al., 2013; Balaine et al., 2013). Although soil freezing and thawing may occur most intensely at the soil surface, it is clear that these soil dynamics will have a significant control on water and gas exchange through the entire soil profile. There needs to be consideration of the effects that GMr incorporation during different times of the year has on biological, chemical, and physical soil properties and processes as they relate to off-season N_2O emissions.

Incorporating GMr with inversion tillage affects soil water infiltration and holding capacity because of a change in the size and distribution of soil pores. Therefore, tillage alters the amount of free water at a particular soil temperature (Parkin et al., 2013), which then can alter

the pattern and extent of soil freezing and thawing dynamics. Thus, when GMr tillage is conducted in the fall, there are physical effects on soil properties and processes that could also affect denitrification and N_2O emissions. First, the exposure of a bare soil surface decreases the soil albedo (Williams et al., 2009) and lessens the ability to trap snow that insulates the soil surface from freezing and thawing events (Iwata et al., 2011), altering thermal and water dynamics known to control N_2O production and the subsequent release to the atmosphere, especially at spring thaw (Henry, 2008; Blankinship and Hart, 2012). Secondly, inversion plowing disrupts preferred pathways (i.e., macropores), which have a major control on water and gas exchange (Carter, 1996; Kay and VandenBygaart, 2002; Abdollahi and Munkholm, 2014). Finally, a temporary (Strudley et al., 2008) reduction in bulk density (Kay and VandenBygaart, 2002) by loosening soil through the tillage depth (Carter, 1996) reduces the occurrence of anaerobic conditions needed for denitrification. Manipulating soil water and surface vegetation with fall tillage is an example of how field management and cropping patterns can decrease or increase freeze–thaw related soil N_2O emissions (Dietzel et al., 2011).

Studies are needed that look at cool season N losses from legumes grown as GMr and forage, with respect to their timing of incorporation (Phillips, 2007; 2008; Clark et al., 2009) because cool season N_2O emissions are not measured or assumed to be minimal in cool soil temperatures (Morkved et al., 2006; Phillips, 2008). Decisions for cropping patterns that include GMr in the rotation should consider how GMr incorporation affects N_2O emissions through the whole year (Aulakh et al., 1991; Rochette and Janzen, 2005; Wagner-Riddle et al., 2007; Brozyna et al., 2013), rather than only taking measurements during the growing season. Furthermore, the effect of GMr management should also consider snow levels that are predicted to diminish in temperate northern latitudes and the subsequent increase in soil freeze/thaw frequency as a result of climate change (Groffman et al., 2001). The objective of this two-year study was to measure soil N_2O emissions (Sep. to June) from red clover GMr that differed in management by the timing of incorporation from early fall to spring. Inorganic N content, CO_2 emissions (e.g., microbial activity), and the effect of air and soil temperatures were also tracked to help interpret the timing and magnitude of N_2O emissions through the off-season.

3.4 MATERIALS AND METHODS

3.4.1 Field site description and experimental design

All field data were collected at the Brookside experimental site for organic agriculture at Dalhousie University's Agricultural Campus, Truro, Nova Scotia, Canada (45°23'24.72" N lat; –63° 15'16.15" W long) from the off-season (Sep.10th–June 1st) over two years. Soil at the Brookside experimental site is a well to imperfectly drained sandy loam (Orthic Humo-Ferric Podzol) from the Woodville soil series (Webb et al., 1991). Soil pH (H_2O) at the beginning of the experiment was measured at 6.3±0.3 and total soil C and N was 19.8±3.9 and 1.7±0.3 kg m^{-2}, respectively, from the top 0 to 15 cm of soil. Mean annual temperature is 5.8°C and annual precipitation is 1224 mm, which includes a mean snowfall of 214 cm. Annual total degree-days (>5°C) average 1700 and the frost-free period is 99 d. Drainage at the site was further improved by the installation in 2008 of subsurface drainage tiles at a depth of 0.8 m.

This study was part of a larger project that is looking at fertility management practices for legumes grown as GMr in red clover (*Trifolium pratense* L.) – spring wheat (*Triticum aestivum* L.) crop sequences within a 4-yr rotation. The crop sequence consisted of soybeans (Year 1), two years of red clover (Years 2, 3), and spring wheat (Year 4). The overall experiment was designed in a randomized block design with four main plots (16 x 75 m) and four sub-plots (16 x 16 m) representing all four phases of the crop rotation. Each treatment and phase was then replicated over three blocks. A diagram of the field plot layout is found in Appendix 1.

We focused on the Year 3 clover phase (GMr) prior to the spring wheat phase (Year 4) of the rotation. Treatments differed in management and timing of GMr incorporation using a moldboard plow (Table 3.1). Treatments included: (1) Clover incorporated in Sep. (Early Fall; EF), (2) Clover incorporated in Nov., with aboveground clover biomass removed (Hayed Late Fall; HLF), (3) Clover incorporated in Nov. (Late Fall; LF), and (4) Clover undisturbed until incorporation in May (Spring; SP).

Table 3.1. Description and timing of GMr treatments at the experimental site 2011–2013.

Treatments	Activity	Year 1	Year 2
Hayed Late Fall	Clover biomass harvested as hay	23 June 2011	18 June 2012
Early Fall, Late Fall, and Spring	Clipped clover, residue remained on the soil surface	23 June 2011	18 June 2012
Hayed Late Fall	Clover biomass harvested as hay	18 Aug. 2011	31 July 2012
Early Fall, Late Fall, and Spring	Clipped clover, residue remained on the soil surface	18 Aug. 2011	31 July 2012
Early Fall	Red clover incorporation	17 Sep. 2011	18 Sep. 2012
Hayed Late Fall and Late Fall	Red clover incorporation	2 Nov. 2011	7 Nov. 2012
Spring	Red clover incorporation	2 May 2012	1 May 2013

3.4.2 Clover biomass and quality

Clover was either clipped (i.e., EF, LF, and SP) or removed (i.e., HLF) twice during the growing season (Table 3.1) to keep clover in its vegetative state. Clover biomass was measured among all treatments prior to clover clipping or removal (HLF only) by weighing a 1.5 m x 16m strip from the center of each plot with a Haldrup forage harvester (J. Haldrup, Løgstor, Denmark) equipped with a biomass scale. A sub-sample was then dried for dry matter (DM) determination and the quality of clover biomass was determined by C and N concentrations using a CN analyzer (Vario Max, Elementar, Hanau, Germany).

3.4.3 Climate, soil temperature and water content monitoring

Soil temperature and volumetric water content were measured using a single 5TE sensor (Decagon Devices, Pullman, WA, USA) in each plot (n=12) for measurements at 15 cm and at the soil surface (0–5 cm) from both plowed and vegetated surfaces in each block. Sensors were installed at the 15 cm depth by excavating a narrow hole with a tree planter's shovel in each plot and inserting perpendicular to the surface in the undisturbed sidewall of the narrow hole. Each sensor had a soil volume of influence of 715 cm^3.

Air temperature and precipitation were measured on-site using a shielded thermocouple (Campbell Scientific, Logan, UT, USA) and tipping bucket with hourly measurements logged to a CR200X data logger (Campbell Scientific, Logan, UT, USA) (Fig. 3.1). Daily mean soil temperatures were calculated from hourly measurements logged to an EM-50 data logger

(Decagon Devices, Pullman, WA, USA). Accumulated degree hours (ΣDH) is the sum of Freezing Degree Hours (ΣFDH) and Thawing Degree Hours (ΣTDH) for air and soil that were calculated by summing the daily degree hours below and above 0°C (Wagner-Riddle et al., 2007; Yanai et al., 2011; Glenn et al., 2012).

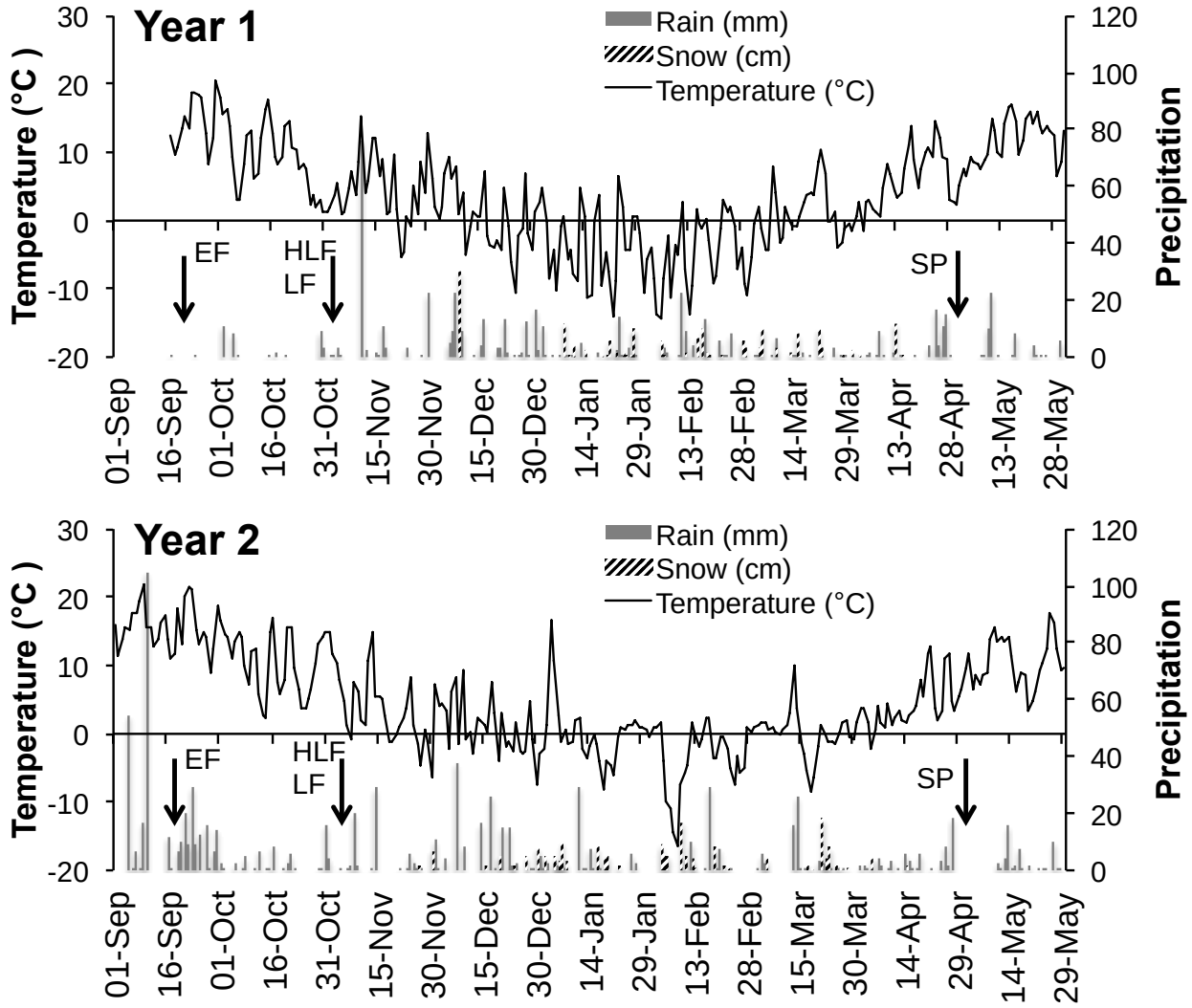

Fig. 3.1. Weather at the Brookside research site in Truro, NS, Canada from Sep. to June over two years. Vertical arrows show the timing of clover incorporation for Early Fall (EF), Hayed Late Fall (HLF) and Late Fall (LF) combined, and Spring (SP).

3.4.4 Soil inorganic N

Surface soil (15 cm) was sampled typically on the same day as gas sampling by compositing 10 soil samples per plot taken with a handheld probe (2 cm i.d.) in Year 1 only. Soil

samples were immediately frozen in an on-site freezer until they were thawed and then extracted for N_{min} with 0.5 M K_2SO_4 (3:1, liquid:soil) for 1-hr and filtered through glass fiber filter paper following the method of Keeney and Nelson (1982) before NH_4^+ and $NO_2^- + NO_3^-$ were analyzed colorimetrically with an auto-analyzer (Technicon Instruments Corporation, Tarrytown, NY, USA).

3.4.5 N_2O and CO_2 surface emissions

A vented, non-steady-state PVC chamber was used for N_2O and CO_2 measurements. Chambers were 20 cm diameter and 15 cm in height, which created a chamber volume of 4.7 L and a surface area of 314.16 cm^2. Reflective and insulated material covered the outside of the chamber. Permanent collars measuring 10 cm in height were placed five to seven cm in the soil and were only removed during field tillage. All plants were removed from inside the collars. There was no change to the gas sampling protocol when collars were filled with snow. However, chambers were placed on top of the snow if snow levels were higher than the collars, which occurred on two sampling dates in Jan. of Year 2.

Emissions of N_2O and CO_2 were measured at least twice per calendar month from Sep.10^{th} to June 1^{st} in Year 1 (2011–12) and Year 2 (2012–13), with the exception of the month of Feb. in Year 1 when collars filled with ice and only a single measurement was taken. An additional sampling date was included in both Jan. and Feb. of Year 2, as well as more frequent sampling after legume residue incorporation in Sep. and Nov. and near the initial thaw periods (8 Mar. 2012 and 14 Mar. 2013). Spring thaw was signaled by the beginning of rapid water infiltration, as measured by on-site tipping buckets used to quantity outflow from tile drains. Emissions of CO_2 from the soil surface are interpreted as heterotrophic respiration (i.e., plant roots and microbes) and used as a proxy for microbial activity, not to assess C cycling and storage in soils. Cumulative N_2O emissions were calculated from Sep. to June in Year 1 (274-d) and Year 2 (273-d) on an individual chamber basis by linear interpolation between sampling dates, using the assumption that gas flux measured on a sample date represents the daily average flux. When a sampling frequency of 14 to 21 d is used, Parkin (2008) reported that these estimates were –43 to 64% of the "true" cumulative emissions, which were estimated with automated chambers four times each day. Flux measurements are dependent on the underlying temporal variability across a particular site during that point in time. Therefore, the temporal

pattern of N_2O–N emissions as a result of differing GMr management is of greater interest than the comparison of cumulative flux between these systems.

Air from the chamber headspace was sampled by withdrawing 20 mL of air through a rubber septum at the top of the chamber at regular intervals using plastic disposable syringes and transferred to 12 mL pre-evacuated (67 pa) vials (Labco Exetainer, High Wycombe, U.K.). A total of 24 chambers (2 sub-samples per plot) were sampled over 4 time intervals (0, 10, 20 and 30 min) for a total of 96 samples per sampling date. Four replicates of 1.1 µL L^{-1} N_2O gas standards were taken on each sampling date to ensure the reliability of the storage and handling of the gas samples (Burton et al., 2008).

Gas analysis was performed using a Varian Star 3800 Gas Chromatograph (Varian, Walnut Creek, CA) fitted with an electron capture detector (ECD), thermal conductivity detector (TCD), and a Combi-PAL Autosampler (CTC Analytics, Zwingen, Switzerland). The Combi-PAL injects 2.5 mL into the gas chromatograph to fill two 0.5-mL sampling loops that load gas onto ECD and TCD/FID flow streams. The ECD was operated at 380°C, 90% Ar, 10% CH_4 carrier gas at 20 mL min^{-1}, Haysep N 80/100 pre-column (0.32 cm diameter×50 cm length) and Haysep D 80/100 mesh analytical columns (0.32 cm diameter×200 cm length) in a column oven operated at 70°C. The pre-column was used in combination with a four-port valve to remove water from samples. The TCD was operated at 130°C, pre-purified He carrier gas at 30 mL min^{-1}, Haysep N 80/100 mesh (0.32 cm diameter ×50 cm length) pre-column followed by a Porapak QS 80/100 mesh (0.32 cm diameter×200 cm length) analytical column maintained at 70°C.

As described by Burton et al. (2008), fluxes (F_c) of N_2O and CO_2 (g N_2O-N ha^{-1} d^{-1} and kg CO_2 ha^{-1} d^{-1}) were calculated using the following equation (Hutchinson and Livingston, 1993):

$$F_c = dC / dt \, VcM_{mol} / (AV_{mol}) \quad [1]$$

where dC/dt is the rate of change in N_2O or CO_2 concentration (mol mol^{-1} h^{-1}), A is the surface area (m^2) of the chamber, Vc is the total volume (L) of the chamber (i.e., chamber plus exposed collar), M_{mol} is the molar mass of N_2O (g mol^{-1}) and V_{mol} is the volume of a mole of N_2O (L mol^{-1}) inside the chamber corrected for temperature using the ideal gas law. The flux value of dC/dt was calculated using the simple linear regression of gas concentrations versus time of the deployment period. The slope of the line multiplied by the chamber volume and

division by the chamber surface area will result in flux units of µL gas m^{-2} min^{-1}. All flux data are presented as g N_2O-N ha^{-1} d^{-1}, or kg CO_2 ha^{-1} d^{-1}.

3.4.6 Statistical analysis

Daily N_2O and CO_2 flux measurements on each sample date were evaluated for Treatment, Time and the interaction of Treatment x Time effects using repeated measured using a mixed model split-plot approach that treated sub-samples (n=6) as the random effect. Monthly cumulative N_2O emissions were evaluated using a similar repeated measures model, where Month was the repeated factor. Cumulative N_2O emissions for the entire season in each year were evaluated for the effect of Treatment using a mixed model where sub-samples was the random effect. Sub-samples were at least 10 m apart in each experimental unit, which allowed us to assume that any spatial auto-correlation would be minimal (Ball et al., 2007). Clover biomass from the first and second clipping were summed together and mean residue quality (i.e., C:N) were tested for differences in Treatment, Year, and their interaction using a mixed model where block was the random factor. The effect of Treatment, Time, and the interaction of Treatment x Time on soil N_{min} in Year 1 only and 15 cm soil ΣDH in both years were analyzed using a mixed model and treating block (n=3) as the random effect. Differences between years for monthly and annual air temperatures were determined with a two-tailed t-test. Treatment means were separated with Tukey's post-hoc test and a test of the simple main effects (slices) when the interaction of Treatment and Time was significant. Soil N_{min}, N_2O, and CO_2 data were log transformed to help satisfy the assumptions of equal variance and normality, while untransformed means and standard errors are presented. All effects were considered significant when $p \leq 0.05$, and statistical tests were performed using JMP software (SAS Institute Inc. Cary, NC, USA).

3.5 RESULTS

3.5.1 Clover biomass and quality

Results in Year 1 refer to field data collected from 1 Sep. 2011 to 31 May 2012 and Year 2 represents the same dates in 2012 and 2013. Prior to the experiment, visual estimates of GMr plant cover in 2nd yr red clover plots confirmed that 90 to 95% of the plot area was covered by red clover, while the remaining plant biomass was a mix of grasses and annual weedy forbs (data not shown). There was no difference in aboveground clover biomass (Table 3.2) across treatments (p= 0.28) or years (0.07), nor was the interaction significant (p= 0.69). Mean aboveground clover biomass at the end of the growing season over the 2-yr study had a range of 8.7 to 10.6 Mg ha^{-1} (dry wt.), which represented an input of organic-N of 238 to 311 kg ha^{-1}. Biomass quality was assessed by C:N ratio and was not different among treatments (p= 0.91), nor was the interaction with Year significant (p= 0.47). The effect of Year was strongly significant (p= 0.0004), yet the biological significance of having small average differences in the C:N ratio GMr tissue (14.2 as compared to 15.6 in Year 1 and 2, respectively) may be considered minor, but does imply a 10% higher GMr biomass N content in Year 1 over Year 2.

Table 3.2. Clover aboveground biomass (Mg ha^{-1} dry wt.) and quality during the growing seasons prior to incorporation in Year 1 and 2.

Treatments	**Biomass (DM Mg ha^{-1})**		**Biomass quality (C:N)**	
	Year 1	Year 2	Year 1	Year 2
Early Fall	9.05 (0.09)†	10.58 (0.36)	14.6 (0.67)	15.6 (0.7)
Hayed Late Fall	8.71 (0.24)	9.00 (0.94)	14.4 (0.68)	15.4 (0.82)
Late Fall	8.77 (0.83)	9.29 (0.57)	14.1 (0.67)	15.4 (1.07)
Spring	9.3 (0.57)	9.93 (0.74)	13.7 (0.53)	16.1 (0.72)
Source of variation	Probability (p)‡			
Treatment (T)	0.279		0.908	
Year (Y)	0.074		**0.0004**	
TxY	0.688		0.472	

† Values in brackets represent the SEM (n=3).
‡ Bolded values indicate a significant difference at the $p \leq 0.05$ level of probability.

3.5.2 N_2O emissions

The Treatment and Time interaction was significant ($p< 0.0001$) in Year 1 (Fig. 3.2 A), and three sample dates between 27 Sep. 2011 and 4 Oct. 2011 all had greater N_2O emissions from the EF treatment as compared to all other treatments. However, on 1 Dec. 2011 the greatest N_2O emissions, although small in magnitude, were measured from LF incorporated clover (3.15 g N_2O-N ha^{-1} d^{-1}) and the smallest from SP plots (0.05 g N_2O-N ha^{-1} d^{-1}). The largest emission event from any treatment was measured on 20 Mar. 2012 from the SP treatment (45.34 g N_2O-N ha^{-1} d^{-1}) and the smallest on this day was measured from the HLF treatment (1.24 g N_2O-N ha^{-1} d^{-1}). Two weeks following SP clover incorporation (16 May 2012), the SP treatment had the largest N_2O emission of 10.8 g N_2O-N ha^{-1} d^{-1} and the least from the EF and HLF (2.45 g N_2O-N ha^{-1} d^{-1}).

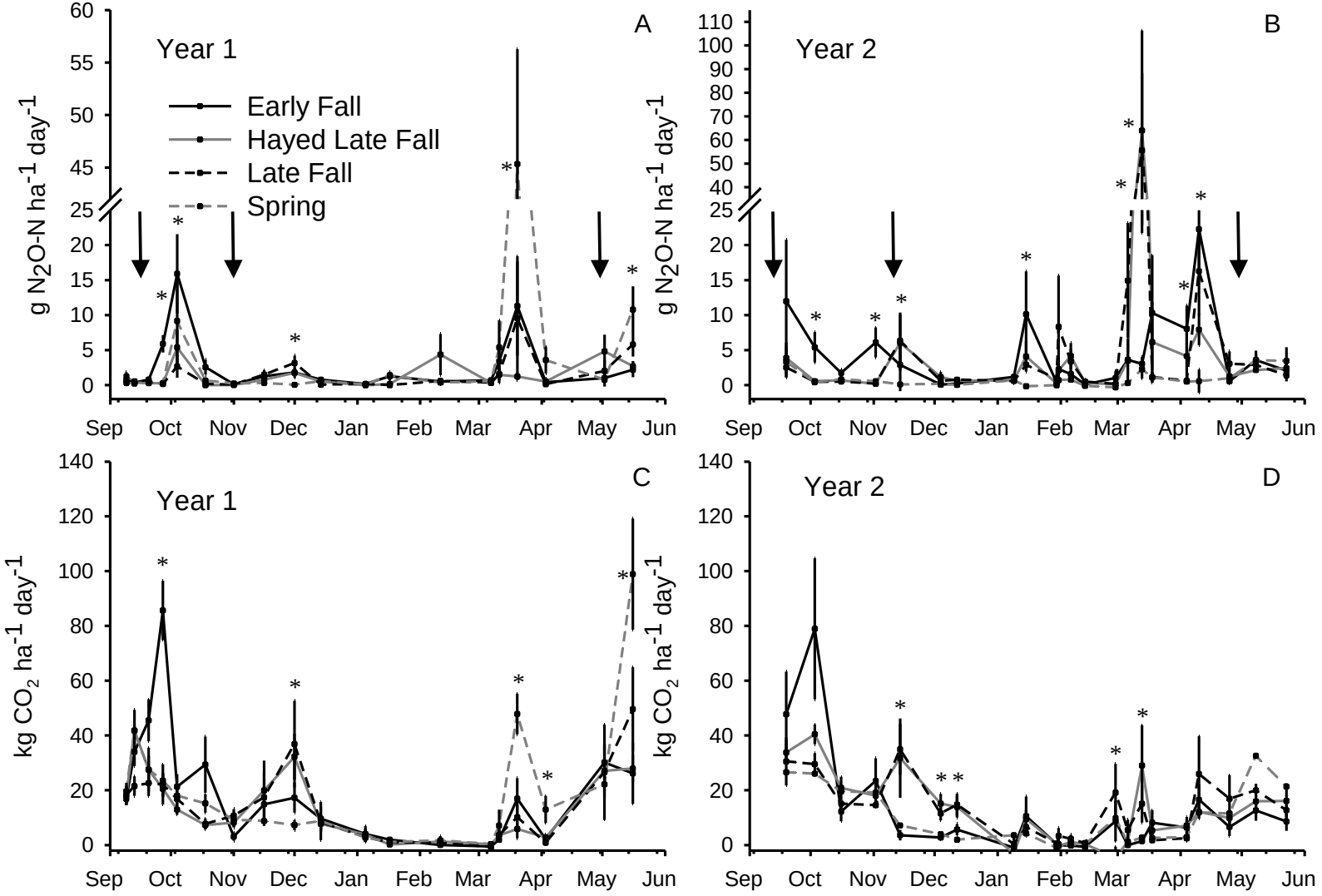

Fig. 3.2. Soil emissions of N_2O (A, B) and CO_2 (C, D) from fall incorporated red clover in Year 1 and 2. Significant differences ($\alpha = 0.05$) in N_2O and CO_2 across treatments for each sample date are indicated with "*". Vertical arrows (from left to right) show the timing of Early Fall (EF), Late Fall (LF) and Hayed Late Fall (HLF) combined, and Spring (SP) clover incorporation in Year 1 and 2. Error bars represent the SEM (n=6).

In Year 2, the Treatment x Time interaction was also significant ($p < 0.0001$) and showed a similar temporal pattern to Year 1 (Fig. 3.1 B). Early Fall clover incorporation produced greater N_2O emissions on 3 Oct. 2012 and 2 Nov. 2012 as compared to all other treatments, while Nov. clover incorporation (7 Nov. 2012) produced the largest N_2O emissions in the LF and HLF treatments (6.19 g N_2O-N ha^{-1} d^{-1}) when compared to SP (0.11 g N_2O-N ha^{-1} d^{-1}). During the middle of winter, the largest emissions were measured from EF (10.12 g N_2O-N ha^{-1} d^{-1}) and the smallest from SP plots (–0.16 g N_2O-N ha^{-1} d^{-1}) on 15 Jan. 2013. The peak of spring thaw in Year 2 was on 13 Mar. 2013 with the greatest N_2O emissions being measured from both the LF and HLF treatments (59.82 g N_2O-N ha^{-1} d^{-1}) as compared to both the EF and SP clover incorporated treatments that had a mean of 2.65 g N_2O-N ha^{-1} d^{-1}. At the beginning of Apr., EF emissions (8.05 g N_2O-N ha^{-1} d^{-1}) were nearly four times emissions from all other treatments (1.75 g N_2O-N ha^{-1} d^{-1}). There was another large emission event on 10 Apr. 2013, when both EF and LF clover incorporation treatments had the largest N_2O emissions with 22.27 and 16.28 g N_2O-N ha^{-1} d^{-1}, respectively, and the smallest emissions were measured from the SP treatment (0.57 g N_2O-N ha^{-1} d^{-1}).

3.5.3 CO_2 emissions

The effect of Treatment across all sample dates (Fig. 3.1 C) was not significant ($p = 0.104$) in Year 1, but the interaction with Time was ($p < 0.0001$). Emissions of CO_2 were larger in EF plots on 27 Sep. 2011 with 85.67 kg CO_2 ha^{-1} d^{-1}, as compared to an average of 22.07 kg CO_2 ha^{-1} d^{-1} from the other three treatments. Both LF and HLF treatments had CO_2 emissions that were 4.7 times as great as compared to SP (7.32 kg CO_2 ha^{-1} d^{-1}) on 1 Dec. 2011, while EF was intermediate (17.31 kg CO_2 ha^{-1} d^{-1}). Undisturbed plots (SP) in Year 1 had the greatest CO_2 emissions (47.89 kg CO_2 ha^{-1} d^{-1}) on 20 Mar. 2012 as compared to both the LF and HLF treatments (8.04 kg CO_2 ha^{-1} d^{-1}), while EF was intermediate with 16.96 kg CO_2 ha^{-1} d^{-1}. Two weeks later on 3 Apr. 2012, SP plots again had 6.5 times the CO_2 emissions as compared to the mean of all other treatments (2.01 kg CO_2 ha^{-1} d^{-1}). On the last sampling date in Year 1 (16 May 2012), CO_2 emissions from SP incorporated clover were again twice as large when compared to LF (49.64 kg CO_2 ha^{-1} d^{-1}), and the smallest from EF and HLF plots (27.08 kg CO_2 ha^{-1} d^{-1}).

The Treatment and Time interaction was also significant ($p = 0.0003$) in Year 2 (Fig. 3.1 D). Between 14 Nov. 2012 and 12 Dec. 2012, the largest CO_2 emissions among treatments were

measured in both LF and HLF treatments, with average emissions across the three sample dates of 20.35 kg CO_2 ha^{-1} d^{-1} when compared to the smallest from both EF and SP (4.2 kg CO_2 ha^{-1} d^{-1}). In contrast to the pattern of CO_2 emissions at spring thaw in Year 1, LF and HLF treatments had emissions that were 11 times greater than both EF and SP treatments (2.14 kg CO_2 ha^{-1} d^{-1}); following the same pattern among treatments for N_2O on 13 Mar. 2013.

3.5.4 Cumulative N_2O

Across all treatments, cumulative N_2O emissions were 48% lower in Year 1 as compared to Year 2 with the highest N_2O emissions (p=0.02) being measured from the SP treatment (950.09 g N_2O-N ha^{-1}) in Year 1, and smallest N_2O emissions from LF and HLF (444.35 g N_2O-N ha^{-1}) treatments (Table 3.3). Early Fall had intermediate values of 599.91 g N_2O-N ha^{-1}. In contrast, the smallest (p=0.047) cumulative N_2O emissions in Year 2 were measured from the SP treatment (261.95 g N_2O-N ha^{-1}) when compared to all three fall incorporated clover treatments (1113.91 g N_2O-N ha^{-1}). When cumulative emissions are analyzed by fall (Sep.–Dec.) and spring (Mar.–June) seasons, delaying GMr incorporation until Nov. did reduce fall N_2O emissions as compared to EF, but emissions did increase in the spring, which effectively canceled out any prior reductions. In Year 1, 50% of the seasonal N_2O emissions from the EF treatment were measured in the fall, as compared to 14, 21, and 24% from the SP, HLF, and LF treatments, respectively. In comparison, EF incorporated clover had 46% of the total N_2O emissions measured in the spring of Year 2, as compared to 65, 70, and 53% from HLF, LF, and SP treatments, respectively.

Table 3.3. Monthly and cumulative N_2O emissions from red clover (Sep.-June) over two years.

Months	Year 1	Treatments				Year 2	Treatments			
		Early Fall	Hayed Late Fall	Late Fall	Spring		Early Fall	Hayed Late Fall	Late Fall	Spring
	$p \leq 0.05$	- - - - - - - g N_2O-N ha^{-1} $month^{-1}$ - - - - - - -				$p \leq 0.05$	- - - - - - - g N_2O-N ha^{-1} $month^{-1}$ - - - - - - -			
Sep	0.019	60.42 a†	14.20 b	12.15 b	14.46 b	0.263	273.82	147.46	64.06	75.80
Oct	<0.001	196.02 a	53.62 b	30.07 b	99.30 b	0.026	120.05 a	18.09 b	14.40 b	21.53 b
Nov	0.089	33.43	25.02	46.13	7.82	0.015	101.42 a	105.30 a	114.76 a	9.55 b
Dec	0.083	27.95	22.50	25.36	12.16	0.24	13.80	26.30	24.05	7.50
Jan	0.466	25.75	22.37	6.40	21.74	0.003	124.47 a	55.66 a	53.93 a	7.00 b
Feb	0.324	18.90	90.49	15.17	19.07	0.005	22.20 a	32.12 a	35.88 a	2.80 b
Mar	0.002	150.88 b	44.31 b	101.61 b	437.00 a	<0.001	201.44 a	537.08 a	440.94 a	30.93 b
Apr	0.252	23.40	72.77	31.42	51.38	<0.001	298.52 a	129.18 a	215.64 a	30.37 b
May	0.131	63.16	129.66	145.46	287.16	0.941	64.79	50.87	55.49	76.47
Cumulative	**0.02**	**599.91 ab**	**474.95 b**	**413.75 b**	**950.09 a**	**0.047**	**1220.52 a**	**1102.07 a**	**1019.15 a**	**261.95 b**

†Means within rows with the same lower case letter for each month and cumulative emissions in each year are not significantly different among treatments at the $p \leq 0.05$ level of probability. Significantly different cumulative emissions are bolded.

3.5.5 Soil inorganic N (Nmin)

Soil N_{min} (0–15 cm) was evaluated using a repeated measures analysis (Fig. 3.2) on samples taken during Year 1 (Sep.–June) only. The overall effect of treatment on NO_3^- (kg NO_3^--N ha^{-1}) was not significant (p= 0.18), yet the Treatment x Time interaction was (p< 0.0001). Late Fall plots had the highest NO_3^- content on 13 Sep. 2011 as compared to the lowest in EF and HLF, while SP was intermediate. One week later (20 Sep. 2011), the highest NO_3^- content was measured from the LF treatment (18.51 kg NO_3^--N ha^{-1}) as compared to all other treatments that had a mean value of 8.57 kg NO_3^--N ha^{-1}. Hayed Late Fall plots had a soil NO_3^- content that was 51% smaller on 27 Sep. 2011 when compared to all other treatments (17.95 kg NO_3^--N ha^{-1}). The lowest NO_3^- contents were again measured from HLF plots (3.98 kg NO_3^--N ha^{-1}) on 18 Oct. 2011, when compared to all other treatments (8.8 kg NO_3^--N ha^{-1}). During spring thaw, the highest NO_3^- content was from LF (37.06 kg NO_3^--N ha^{-1}), followed by HLF (25.27 kg NO_3^--N ha^{-1}), as compared to both EF and SP treatments (14.42 kg NO_3^--N ha^{-1}).

The content of NH_4^+ at the soil surface (0–15 cm) over all sampling times was significantly greater (p= 0.04) from the SP treatment (9.15 kg NH_4^+-N ha^{-1}), as compared to the least in EF (5.8 kg NH_4^+-N ha^{-1}). On 7 Jan. 2012, EF had a NH_4^+ content (kg NH_4^+-N ha^{-1}) that was nearly 2.5 times lower than all other plots with a mean of 10.72 kg NH_4^+-N ha^{-1}. At the end of the study period (23 Apr. 2012), EF had the lowest NH_4^+ content (2.08 kg NH_4^+-N ha^{-1}) when compared to all other treatments (4.64 kg NH_4^+-N ha^{-1}). Following SP clover incorporation on 2 May 2012, NH_4^+ in SP incorporated clover plots went from 13.62 to 23.53 on 16 and 24 May, respectively which was 68 and 181% greater than all other treatments.

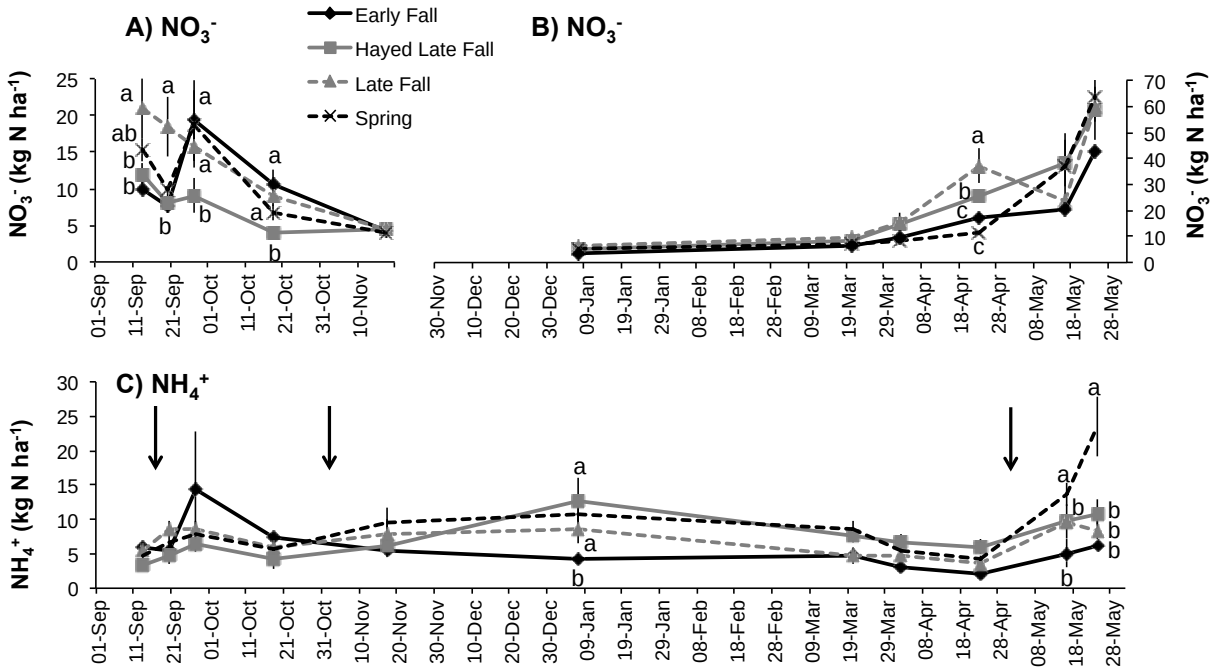

Fig. 3.3. Soil NO_3^- (A, B) and NH_4^+ (C) content from the 15 cm soil depth from Sep. to June in Year 1. NO_3^- is divided into 2 panels due to the different scale used from Sep. to early Nov. (A) and late Nov. through May (B). Arrows represent the timing of clover incorporation (from left to right) for Early Fall (EF), Late and Hayed Late Fall (LF, HLF), and Spring (SP) treatments. Means with the same lower case letter are not significantly different at the $p \leq 0.05$ level of probability. Error bars represent the SEM (n=3).

3.5.6 Air temperature

A two-tailed *t*-test revealed no significant difference (p=0.16) in the seasonal (Dec. to Apr.) average air temperature between the two years of study. Mean air temperatures were –1.9°C for Year 1 and –2.8°C during Year 2 (Table 3.4). Although the small change in average temperature seems insignificant, the difference in temperature was actually a result of 1582 more freezing hours (ΣFDH) in Year 2, and 1294 less overall thawing hours (ΣTDH). December mean air temperatures were on average 1°C warmer in Year 1, as compared to Year 2, yet not different (p=0.16). However, the sum of freezing and thawing hours (ΣDH) produced a positive value in Year 1 (448) and a negative value (–313) in Year 2. Air temperatures were significantly (p= 0.05) lower in Jan. of Year 2 (–6.8°C), as compared to Year 1 (–3.7°C). Mean temperatures in Feb. were not different (p= 0.6), which was also reflected by similar warming and cooling degree hours. During the month of Mar., mean air temperatures were not different (p= 0.77), but temperature means and associated accumulated degree hours (ΣDH) during the first half of Mar.

revealed considerably more freezing conditions in Year 1. In contrast, there were more warming conditions (ΣDH) from Mar. 16 to 31 in Year 1 as compared to Year 2.

Table 3.4. Average daily air temperature from December to March in Year 1 and 2, Freezing Degree Hours (ΣFDH), Thawing degree hours (ΣTDH), and their sum (ΣDH).

Dates	Mean Daily Air Temperature		$p \leq 0.05$	Air Freezing Degree Hours (ΣFDH)		Air Thawing Degree Hours (ΣTDH)		(ΣDH)† = (ΣFDH) + (ΣTDH)	
	Year 1	Year 2		Year 1	Year 2	Year 1	Year 2	Year 1	Year 2
Dec‡	**0.6**	**–0.4**	**0.16**	**–1445**	**–1819**	**1893**	**1506**	**448**	**–313**
1–15	2.8	0.5		–270	–830	1290	1005	1020	175
16–31	–1.5	–1.3		–1175	–989	603	501	–572	–488
Jan	**–3.7**	**–6.8**	**0.05**	**–3535**	**–5673**	**812**	**640**	**–2723**	**–5033**
1–15	–3.3	–5.4		–1595	–2171	412	210	–1183	–1961
16–31	–4.0	–8.0		–1941	–3502	399	430	–1542	–3072
Feb	**–4.9**	**–4.2**	**0.6**	**–3532**	**–3242**	**370**	**325**	**–3162**	**–2917**
1–14	–7.5	–6.4		–2614	–2202	109	65	–2505	–2137
15–28	–2.0	–2.3		–918	–1040	262	260	–656	–780
Mar	**0.4**	**0.2**	**0.77**	**–1794**	**–1154**	**1956**	**1266**	**162**	**112**
1–15	–1.1	1.6		–1219	–185	659	775	–560	590
16–31	1.8	–1.2		–575	–969	1297	491	722	–478
Total§	**–1.9**	**–2.8**	**0.37**	**–10306**	**–11888**	**5031**	**3737**	**–5275**	**–8151**

† The sum of freezing and thawing hours (ΣDH)
‡ Monthly and total temperatures means for Year 1 and 2 are bolded and bi-weekly means are below.
§ Total refers to the seasonal (Dec.–Apr.) average temperature.

3.5.7 Soil temperature

Accumulated freezing hours were evaluated twice monthly for differences among treatments at the 15 cm soil depth (Fig. 3.3). During the first year of study, there were no differences in ΣFDH among treatments (p= 0.7), nor was the interaction with Time significant (p= 0.74). In Year 2 there was nearly four times the number of freezing hours in LF and HLF plots, and twice as many in the EF treatment as compared to SP, yet overall was marginally significant (p= 0.07). In the second half of Jan. there were 713 more freezing hours recorded in LF and HLF plots, and 344 more in EF as compared to SP. At the beginning of Feb. there were 255 more freezing hours in the three fall incorporated clover plots as compared to SP.

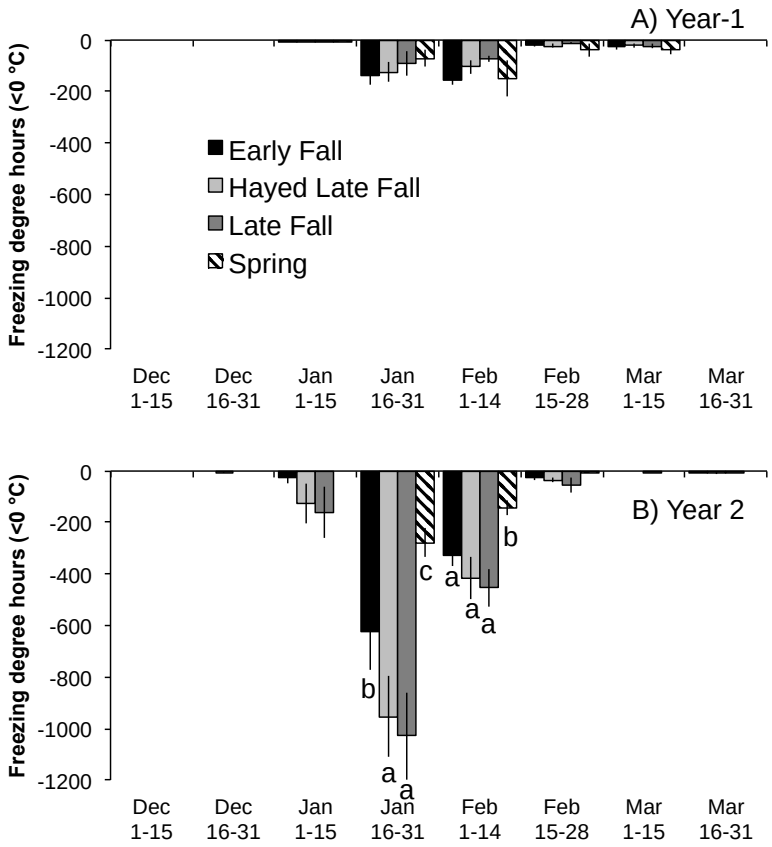

Fig. 3.4. Freezing degree hours (ΣFDH) from the 15 cm soil depth during Year 1 (A) and Year 2 (B). Treatments with the same letter are not significantly different at the $p \leq 0.05$ level of probability. Error bars represent the SEM (n=3).

3.5.8 Freeze thaw effects on the timing of N_2O emissions

The rapid increase in soil temperature and water filled pore space (WFPS) at the 15 cm soil depth during spring thaw in March (Fig. 3.4) occurred as the highest daily N_2O emissions are measured from the soil surface in both years of study. The increase in soil temperature at 15 cm is likely a result of both convection of solar energy from the soil surface and the mass transfer of heat from water at the soil surface, an indication that soil water infiltration is no longer restricted by ice. Soil temperatures remained at or near 0°C in all treatments through the beginning of Mar., which is a signal that soil water is in the process of making a complete phase change to liquid (Outcalt et al., 1990). During this time, macropores (> 75 µm) that are largely responsible for water and gas exchange through the profile start to melt before smaller pores and rapid infiltration begins. The higher WFPS in the HLF treatment (Fig. 3.4 E) at the peak emission event that was measured in all other treatments in Year 1 (Fig. 3.4 G) is evidence of N_2O

reduction to N_2, which was more likely from natural field variability, than an effect of treatment because the only change in this treatment, as compared to LF, was the removal of aboveground biomass in two cute of hay from HLF the previous summer, which does not adequately explain the rapid and greater increase in soil water content and the observed decline in N_2O emissions.

Surface soil temperatures (0–5 cm) were measured from both bare soil (fall tilled treatments) and a vegetated surface (spring plowed clover; SP). Sensors used to measure surface temperatures had a variable measurement depth due to the frequent action of freezing and thawing and the production of ice lenses that moved the sensor through the measurement depth (0–5 cm), but are still a good measurement of surface temperature dynamics between treatments. An increase in the magnitude and frequency of daily air freezing and thawing cycles (Fig 3.1) created similar surface conditions between the two cover types in Year 1 and surface soil temperatures did not vary through spring thaw. In contrast, variation in surface temperatures between bare soil (fall plowed) and vegetated cover (spring plowed) through the thawing period is likely due to the observed greater snow trapping and the higher albedo of vegetation as compared to bare soil.

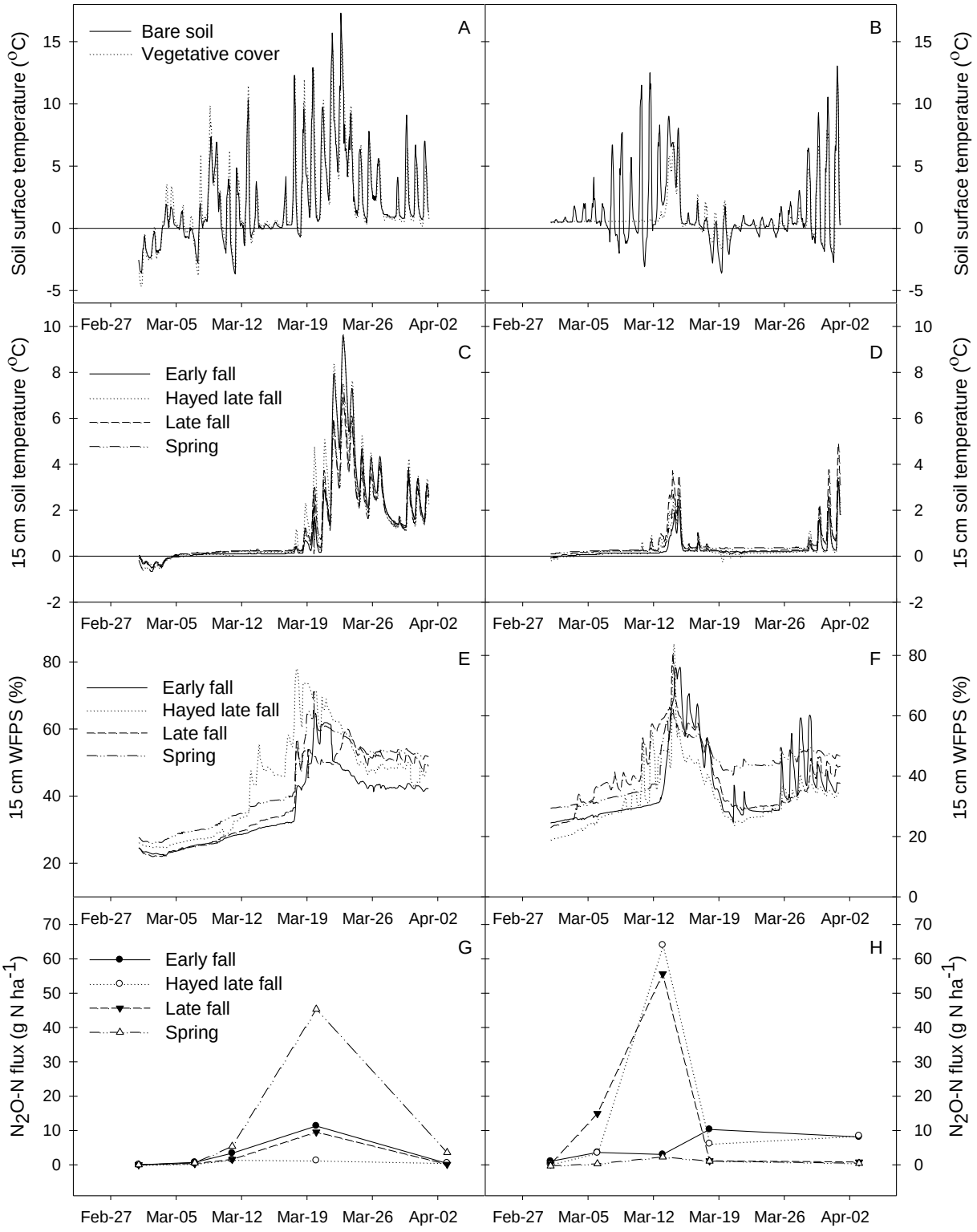

Fig. 3.5. The timing of N_2O emissions during "spring thaw" (Feb.–Apr.) Years-1 and -2 with soil surface temperature (A, B), soil temperature at 15 cm soil depth of (C, D), water filled pore space (WFPS) at 15 cm soil depth (E, F), and N_2O flux (G, H). Surface temperature is from the 0–5 cm soil depth in bare (fall plowed) and cover (Spring; SP) plots.

3.6 DISCUSSION

3.6.1 Timing of GMr incorporation and N_2O emissions

We measured a large amount of year-to year variation in cumulative off-season (Sep.–June) N_2O emissions with 48% more N_2O emissions in Year 2, as compared to Year 1 and a range of 0.26 to 1.22 kg N_2O-N ha^{-1} between all treatments and years. However, the timing and proportion of off-season emissions was consistent over the two years; 60% of cumulative off-season emissions (Sep.–June) were measured in the spring following spring thaw (Mar.–June) and 26 to 31% in the fall (Sep.–Dec.). An exception to this overall trend was the EF treatment that had an average of 65 and 66% greater fall emissions than all other treatments in Year 1 and 2, respectively. Katayangi and Hatano (2012) compared winter to annual cumulative N_2O emissions from croplands in 13 studies without specifying the winter measurement period in each study and determined that winter N_2O emissions can range from 0 to 19 kg N ha^{-1} and be anywhere from 0 to 93% of all annual emissions. Using micrometeorological techniques, Wagner-Riddle et al. (2007) reported off-season N_2O emissions (Nov.–Apr.) over five different years and two management regimes of 0.42 and 2.9 kg N ha^{-1}, as compared to 0.08 to 0.89 kg N ha^{-1} over the same measurement period in our study. Typically, peak off-season emissions are attributed to the actions of soil freezing and thawing (e.g., Matzner and Borken, 2008), or melting snow (Risk et al., 2013), and have been collectively referred to as freeze–thaw related emissions. In our study, the greatest daily flux events occurred during the month of Mar., but this was not consistent in all treatments; spring thaw emissions in Mar. ranged from 9 to 49% of the cumulative off-season (Sep.–June) emissions across all treatments and years.

The biggest difference in N_2O emissions at spring thaw when compared over two years was in the SP treatment. Spring incorporated clover differed from all three fall incorporated clover treatments in both years, although in an unexpected direction in Year 1. Spring incorporation had N_2O emissions that were 91% larger (p= 0.02) than the three fall incorporated treatments in Year 1, but in Year 2, N_2O emissions were only 24% of the three fall treatments.

Baggs et al. (2000) found SP incorporation of a 2-yr-old grass–white clover sward produced 0.24 kg N_2O-N ha^{-1} from Mar. to June, which is less than the emissions measured in our study, but comparable. Wagner-Riddle et al. (2007) reported 32 to 48% greater off-season (Nov.–Apr.) N_2O emissions in fall tilled as compared to no-till in three of the five study years and attributed these differences to greater insulation at the soil surface from crop residues at the soil surface in no-till plots and less soil freezing. Results in Year 2 of our study were similar to Wagner-Riddle et al. (2007), but not in Year 1.

Over the entire off-season period, there were no differences in cumulative N_2O emissions from any of the three fall incorporated clover treatments in either year. However, the timing of these emissions differed and occurred as bursts of N_2O immediately after incorporation for the EF treatment (Sep.–Oct.) of both years, and an increase following Nov. incorporated treatments in Year 2 (Table 3.2). Similar studies that investigated off-season N_2O emissions from the timing of fall GMr incorporation are rare; however, it is not uncommon for N_2O emissions to increase following tillage (Mutegi et al., 2010), fall manure application (Rochette et al., 2004), or alfalfa plowdown in Sep. (Wagner-Riddle and Thurtell, 1998). It was surprising that the removal of roughly 9 Mg ha^{-1} as dry above-ground clover biomass in the HLF treatment did not significantly change cumulative N_2O emissions when compared to LF, or EF treatments. There are a few conclusions that can be drawn from this result: (1) a large proportion of clover N is retained within the root and crown portions of the plant, (2) when clover residue is mulched at the soil surface, rather than removed as hay, much of the N is lost, or (3) N_2O emissions are limited by other factors than just the availability of substrate (i.e., placement, drainage, temperature effects on enzyme and microbe activity). However, it was discussed earlier that the reduced spring thaw emissions were as a result of reduced O_2 conditions, as indicated by the increases in WFPS in Year 1 during spring thaw (Fig. 3.4 E) and the subsequent, likely reduction of N_2O to N_2 needs to be considered when interpreting these results.

The measurement of root and microbial respiration (i.e., CO_2 flux) following clover incorporation is a good indicator of residue decomposition and microbial activity. These measurements provide some insight on the timing of the microbial N demand and the supply of substrate (i.e., NO_3^- and labile C) needed during denitrification. Increases in N_2O emissions were linked to CO_2 emissions, especially following clover incorporation in both years and when clover residue was left on the soil surface (SP) at spring thaw in Year 1. While increased microbial

activity and the possibility for limited O_2 availability as a result can promote denitrification (Burton et al., 2012), it is important to note that the duration and distribution of anaerobic conditions at this site would be dramatically reduced due to sub-surface drainage tiles that would likely limit prolonged denitrification production, and ultimately N_2O emissions (Ball et al., 2008). Drain tiles would also encourage the downward movement of N_2O dissolved in water (Chapter 4), as N_2O solubility in water increases exponentially with decreasing temperature (Davidson and Firestone, 1988; Heincke and Kaupenjohann, 1999; Roper et al., 2013). Considering the improved drainage conditions at the site and the year-to-year variability among treatments, total N_2O emissions are relatively low in both years, but not insignificant. The necessity of full season measurements, in addition to multiple years of measurements across varied management scenarios, is highlighted by these results.

3.6.2 Soil properties and N_2O emissions

It is unclear what amount of N is released from clover residues when mulched at the soil surface and then recycled back into the rhizosphere versus being lost from the system altogether. However, from soil NO_3^- measurements, we can see that the removal of above-ground legume residue in the HLF treatment reduced NO_3^- in bulk soil through the early fall in Year 1 when compared to all other treatments (Fig. 3.2). Early Fall and SP had lower NO_3^- contents when compared to LF and HLF plots in late Apr., with LF being greater than HLF. This indicates that there is substantial N stored below ground and the timing of incorporation in early Nov. is an effective way to slow N mineralization from incorporated GMr through the winter, but mineralization and nitrification may increase in early spring before cash crop planting, which poses a significant risk for increased environmental N losses.

Increasing NH_4^+ concentrations (net mineralization) in cold soils has been reported (Malhi and Nyborg, 1986; Cookson et al., 2002; Smith et al. 2010), which would be in agreement with our finding in Year 1 (Fig. 3.2). Significantly less NH_4^+ was present in the EF plots when compared to all other treatments at the beginning of Jan.; indicating that the easily decomposed and mineralized substrate was depleted quickly soon after incorporation in Sep. (EF). Immobilization and nitrification has been suggested to be more sensitive in colder (e.g., 2–5°C) soil temperatures (Cookson et al., 2002; Hoyle et al., 2006; Clark et al., 2009), while another study reported little difference in these N dynamics when rye grass was incubated at 3°C

(Korsaeth et al., 2002). Cookson et al. (2002) proposed that heterotrophic organisms may be more competitive for NH_4^+ than autotrophic nitrifiers in cold soils when sufficient C substrate is present, which would favor immobilization, rather than nitrification. Increased NH_4^+ concentrations in early Jan. in the SP, HLF, and LF without a parallel increase in NO_3^- suggests that mineralization was not limited in cold soils, but nitrification may have been slower at these temperatures, as suggested above.

The pattern of increased N_2O emissions following clover incorporation was mirrored by increased CO_2 emissions. Microbial and plant respiration are responsible for the production of CO_2, therefore, it is not surprising that we measured increased CO_2 emissions following an increase in substrate availability when clover was incorporated. Cumulative CO_2 emissions have been used to help explain total N_2O emissions (Snowdon et al., 2013); CO_2 production is both an indicator of C availability to soil organisms that may be responsible for N_2O production as well as the consumption of soil O_2, which creates the conditions needed for denitrification. The increase in substrate availability following GMr incorporation together with an excess of soil water from frequent fall precipitation, as occurred especially in Year 2 (Fig. 3.1), creates ideal conditions for N_2O production through denitrification (Petersen et al., 2013). Moreover, N_2O production would increase when a growing crop is lacking following fall GMr incorporation from greater NO_3^- availability (Miller et al., 2008). When a regression of cumulative N_2O (Sep.–June) against cumulative CO_2 (Sep.–June) was performed on our 2-y data set (data not shown), we do indeed see a weakly positive, yet non-significant relationship (R^2= 0.37; p= 0.11; n=8). The relationships of cumulative CO_2 and N_2O as discussed above are primarily made in studies that have focused on the growing season, as opposed to our study that has concentrated on the off-season. Thus, it is not surprising that our positive relationship with CO_2 and N_2O is not as strong as those that have been previously documented.

3.6.3 Effects of soil freezing and thawing on N_2O emissions at spring thaw

The length, duration, and frequency of freezing and thawing events and the availability of substrate are important drivers of spring thaw N_2O emissions, yet it is difficult to predict which mechanism has the greatest influence. Risk et al. (2014) described spring thaw emissions as having two peaks; the first being from N_2O that had accumulated within the soil profile and released upon soil thawing and the second being produced and emitted at the soil surface (*de*

novo), with the latter being larger than the first. Much of the current research suggests that the majority of N_2O emissions associated with soil freezing and thawing are produced close to the soil surface (Rover et al., 1998; Wagner-Riddle et al., 2010; Maas et a., 2013). However, we would have expected to see more N_2O emissions earlier in Mar. from *de novo* N_2O production at the soil surface, as surface soil (0–5 cm) was experiencing complete thawing and then re-freeze on a diel basis. In both years, surface soil temperatures were fluctuating from near freezing up to 10°C on a daily basis for at least a week before the major N_2O emission event occurred during thaw (Fig. 3.4), while temperatures deeper in the soil profile remained close to 0ºC.

The link between soil freezing degree hours (ΣFDH) at 15 cm and N_2O emissions helped to explain the difference in emissions among treatments in Year 2 of our study, but soil freezing was shallow in Year 1 and no clear relationship with freezing and N_2O at spring thaw could be made (Fig 3.3). In contrast to soil freezing being a driver of spring thaw N_2O emissions, the rate and timing of soil thawing could also have a significant control on these emission patterns. In the first year of study, a rapid increase in air temperature in the second half of Mar., as indicated by 1297 ΣTDH (Table 3.4), triggered the complete thawing of the soil profile and the subsequent release of N_2O at the soil surface. In comparison, the entire month of Mar. had 1266 ΣTDH in Year 2, as compared to 1956 ΣTDH in Year 1. Soil temperature, and ultimately the depth of freezing, mirrored the fluctuating air temperature and snow level at the surface. A vegetated surface through the winter will trap more snow and provide greater insulation to freezing air temperatures (Williams et al., 2009; Parkin et al., 2013), and is likely why there was greater freezing at the surface in the fall tilled plots, as compared to the vegetated surface of the SP treatment in Year 2 (Fig. 3.3). The frequent freeze–thaw events through the winter in Year 1—as indicated by the fluctuating air temperature in Fig. 3.1 and ΣDH in Table 3.4—likely eliminated the potential for prolonged soil freezing and could be used as an explanation for why treatment differences were not observed in Year 1. This was in contrast to surface and air temperatures in Year 2 that produced two sequential freeze events that allowed the soil to freeze quickly and deeply at the end of Jan. and into Feb. (Fig. 3.2); as indicated by 3532 and 5673 ΣFDH in Year 1 and 2, respectively, while ΣTDH was similar between years. Interpreting year to year variation in average air temperature and its subsequent effect on soil freezing and thawing patterns is challenging because of the amount of daily fluctuations. However, quantifying the number of

hours below and above 0°C using ΣDH does provide a clearer picture of the influence of weather on soil processes.

There was a significant difference in total off-season N_2O emissions from SP clover incorporation as compared to all fall tillage treatments in both years of study, yet in opposite directions in each year, suggesting that weather also has a strong influence on N_2O emissions. Emissions from SP in Year 1 were 1.9, 3.3, and 8.9 times greater than EF, LF, and HLF during the month of Mar., respectively, which would suggest that either denitrification was greater in SP, or N_2O was further reduced to N_2 in the fall tillage plots before being emitted at the soil surface. Rather than being tied to freezing characteristics, N_2O emissions measured from the undisturbed plots (SP) in Year 1 were more likely caused by the rapid increase in air and soil temperature and the availability of substrate directly at the soil surface, which is consistent with other studies (Ball et al., 1999; Ball, 2013). The improved drainage characteristics in the SP plots, as compared to the fall tillage treatments that would have lacked the continuity of soil pores through the profile, may have reduced WFPS slightly that favored the emission of N_2O, rather than N_2 in Year 1. Improved drainage in SP could also be used as a possible explanation for the low N_2O emissions in Year 2, when compared to the three treatments with fall tillage that had N_2O emissions in Mar. that were 5.5, 13.3, and 16.3 times greater than SP in the EF, LF, and HLF treatments, respectively.

Variation in year to year N_2O emissions across treatments is not surprising when you consider how GMr management systems, such as those in this study, can alter substrate and O_2 supply in soil freezing and thawing conditions from a modification of the bio-physical soil environment system following tillage. Smith et al. (2003) reviewed the interactions of soil physical properties and biological processes and suggested that the production and consumption of N_2O are biological processes, while the release of N_2O is a factor of physical soil properties. As evidenced in the temporal pattern of soil ΣFDH at the 15 cm soil depth (Fig. 3.3), the duration and depth of soil ice can be dramatically different from year to year and was a factor of snow depth, which was altered by the presence of crop residue and the timing of fall plowing. The formation of soil ice is highly influenced by soil water content and pore size distribution, which would clearly be modified by fall-tillage, and thus, help to explain why the timing of plowing altered soil-freezing dynamics. Some of the discrepancy in water infiltration from soil freezing and thawing could be from the disruption of soil aggregates (Dagesse, 2013), or macropore

connectivity (Strudley et al., 2008), as freezing and plowing can lead to a decrease in pore size and connectivity (Bullock et al., 1988). A change in the size of soil pores and their connectivity (i.e., tortuosity) between tilled and no-till soils and the concomitant influence on soil freezing and thawing dynamics is an example of how similar soils, even when located in the same field, can have different soil freezing and thawing dynamics (Grant et al., 1990; Parkin et al., 2013).

What was clear from our study was the increase of soil temperature at 15 cm and the subsequent N_2O emission event; indicating that soil thawing through the soil profile had a control on the timing of the N_2O burst at spring thaw. The parallel increase in water content and temperature at 15 cm measured in this study was in agreement with Wagner-Riddle et al. (2007) and Mass et al. (2013) who measured peak emissions at spring thaw when soil temperature and water content were increasing together at the soil surface. However, we observed that this relationship was not only occurring at the soil surface, but also at 15 cm, which would suggest that hyrdrological properties deeper in the soil profile had a more significant control on the timing of N_2O emissions in our study. In our study, peak N_2O emissions at spring thaw were timed very closely to complete thawing of the soil profile and the rapid movement of soil water, greater WFPS, and an increase in soil temperature (Fig. 3.4). The timing of N_2O emissions at spring thaw and the increase in soil water drainage indicates a link between the hydrology at the site and gas exchange at spring thaw, which has been suggested to be a major control on N_2O emissions (Castellano et al., 2010). Temperature dynamics and the resulting effect on water and gas exchange through the plow depth clearly have a tight control on the timing and magnitude of N_2O emissions. This could help explain why we measured contrasting emission patterns among GMr treatments during Mar. soil thawing.

As climate change models commonly predict a decrease in winter temperatures and an increase in the frequency of freeze–thaw events (Makoto et al., 2013), snow levels in some temperate northern latitudes will undoubtedly decrease. While a snow reduction could increase soil freezing and subsequent N_2O emissions, as seen in Year 2 of our study, there is also the possibility for soil freezing to be concentrated at the soil surface with the increase in freeze–thaw events, which could limit N_2O emissions, as seen in Year 1. A snow removal experiment in a cover-cropped field in New York indicated greater minimum temperatures, but only limited evidence of increased N_2O emissions from fallow plots that experienced greater soil freezing when snow was removed (Dietzel et al., 2011). Wagner-Riddle et al. (2007) measured a

difference in soil temperatures (5 cm) between bare (–3.3°C) and vegetated surfaces (+0.1°C) and differences in N_2O emissions (Nov.–Apr.) between treatments were calculated as being 76% higher from bare soil (2.9 kg N_2O ha^{-1}) as compared to those measured from a vegetated surface (0.7 kg N_2O ha^{-1}). However, as indicated in our study, reduced snow levels can be as a result of frequent freeze–thaw events that reduce prolonged and deep soil freezing and ultimately the number of soil freezing days (Henry, 2008). Some of the predictions of increased soil freezing with a reduced snow pack are a result of snow removal experiments, which may not be a realistic simulation of actual soil freezing dynamics as a result of weather that causes a reduction in snow levels.

3.7 Conclusions

It is clear from this two-year study that there is potential for considerable N_2O emissions from perennial legume GMr crops, regardless of the timing or season of incorporation. Delaying GMr incorporation until spring did reduce N_2O emissions in one of our study years, but also produced the highest emissions at spring thaw in the other year. Early Fall GMr incorporation led to greater emissions in the fall, while delaying GMr incorporation until Nov. (i.e., LF, HLF) only proved to shift the season of emissions until after the winter. The removal of GMr biomass in the HLF treatment did not reduce overall N_2O emissions, which followed a similar temporal emission pattern as LF; indicating that there is considerable N located belowground in a 2-yr old legume GMr crop that becomes susceptible to off-season N losses as N_2O. The use of aboveground legume biomass for forage may be a more efficient use of N, as there was no significant increase in N_2O emissions when this biomass was soil incorporated. However, it should be noted that there was also an increase in WFPS during thaw in Year in HLF plots, which may have led to the consumption of N_2O, rather than a decrease in production.

The lack of a clear relationship with the length and duration of soil freezing, as measured by ΣFDH, and the emission of N_2O at spring thaw was surprising. However, when air temperature was interpreted using both cumulative freezing and thawing hours this information was helpful in interpreting seasonal differences between years, which were used in interpreting gas emissions at spring thaw. The hydrological influence on spring thaw emissions were evident in both years of study and emphasized the importance of understanding the belowground

dynamics of soil freezing, gas accumulation, and subsequent soil thawing and gas emissions from the soil surface.

4. WINTER SOIL PROFILE N_2O CONCENTRATIONS AND DISSOLVED N_2O IN TILE DRAINAGE WATER WITH VARIED TIMING OF RED CLOVER TILLAGE

4.1 Preface

The goal of Chapter 3 was to analyze off-season (Sep.-June) surface N_2O emissions as affected by the timing of red clover GMr tillage and weather over two years. The focus in Chapter 4 is on belowground dynamics of N_2O and CO_2 during those emission events. The physical release of N_2O stored below a frozen soil layer is one mechanism that contributes to elevated N_2O emissions at spring thaw. Much of the previous work in this area has focused on N_2O diffusion to the atmosphere, with little discussion of the possibility of its downward movement in the soil profile with water. In Atlantic Canada, the installation of sub-surface drainage improves soil conditions and extends the limited growing season—especially in the spring when snowmelt and high precipitation can make field accessibility impossible. However, discharging soil water that is supersaturated with N_2O is a pathway for N_2O emissions that may be overlooked. Dissolved N_2O in drainage water was collected from plots that included all phases of the 4-yr crop rotation throughout the study period.

4.2 Abstract

Emissions of N_2O at spring thaw are caused by the rapid production of N_2O and the physical release of previously produced N_2O under frozen soil as thawing begins, yet the downward movement and the potential for further emissions to the atmosphere at sites with drainage tiles installed has rarely been considered. Concentrations of N_2O within the soil profile (7.5–57.5 cm) from Dec. to May over two years were examined in an experiment with varied season and timing of GMr incorporation to examine how the temporal dynamics of belowground N_2O relate to N_2O emissions at the soil surface and N_2O dissolved in drainage water. Treatments included: GMr incorporated in Sep. (Early Fall; EF), Nov. GMr incorporation with aboveground biomass removed during the growing season (Hayed Late Fall; HLF), GMr incorporated in Nov. (Late Fall; LF), and undisturbed GMr that was incorporated in May (SP). In Year 1, Soil N_2O concentrations reached a maximum at the 15 and 30 cm depths in the SP and EF treatments on 8 Mar. 2012 with a mean (±SD) of 120±25 µL L^{-1}, while the highest concentrations at 50 cm were measured on 11 Mar. 2012 from EF (109 µL L^{-1}). During spring thaw in Year 2, SP produced the lowest N_2O concentrations at all depths, while all other treatments had a mean of 98±20 µL L^{-1} (15 cm) and 113±37 µL L^{-1} at 30 to 50 cm. Following extensive drainage, soil concentrations at all depths declined to background levels (0.3 µL L^{-1}), but the decline was more rapid in Year 2 due to a 54% increase in drainage water. There was no difference in dissolved N_2O (µg N_2O-N L^{-1}) in tile drain flow among treatments in Year 1 or 2 (p= 0.7–0.97), yet increased during spring thaw in all treatments with maximum values of 167 to 269 µg N_2O-N L^{-1}. Overall losses of dissolved N_2O (g N_2O-N ha^{-1}) were 14 and 18% of N_2O emitted from the soil surface during the same period and are therefore a significant pathway for N_2O losses.

4.3 Introduction

Emissions of N_2O from the agricultural sector are a substantial component of global N_2O emissions (Janzen, 1998; Butterbach-Bahl et al., 2004; Ambus et al., 2011), yet despite decades of research devoted to the processes that contribute to these emissions there is less of an understanding on the effects of cropping practices on off-season emissions in temperate climates when compared to in-season emissions. Canadian off-season N_2O emissions were estimated to

be 30% of total annual emissions from cropland (Smith et al., 2004), although a much greater proportion of emissions have been reported in eastern Canada (i.e., 30–90%; Wagner-Riddle et al., 2007), as well as emissions from northern Europe (e.g., Rover et al., 1998; Syvasalo et al., 2004). Off-season emissions are typically associated with the actions of freezing and thawing soils, as such, the majority of these emissions occur as brief, yet vigorous events during spring thaw (Christensen and Tiedje, 1990; van Bochove et al., 2000b; Jacinthe et al., 2002; Wagner-Riddle et al., 2007).

Because soil surface N_2O emissions are a result of the complex interactions of soil biological, physical, and chemical properties within the soil profile and at the soil surface (Smith et al., 2003; Castellano et al., 2010; Ball et al., 2013), it can be difficult to pinpoint how management decisions affect soil processes responsible for these emissions. The season and timing of GMr incorporation has not been thoroughly evaluated in the field with respect to freeze–thaw related N_2O emissions, however a legume forage pasture in Germany was shown to have multiple daily N_2O flux rates of 50 to 100 g N_2O-N ha^{-1} d^{-1} that were associated with soil freeze–thaw cycles, as well as events that followed legume incorporation at the end of the three year forage cycle (Kaiser et al., 1998). When soybean residues were incorporated in a laboratory experiment, N_2O emissions during 4-d incubation at 1ºC had a four-fold increase in a silty clay soil when compared to a sandy loam, which was only one example of the intricate interactions with soil type, crop residue addition, and the length and magnitude of freezing on the dynamics of N_2O emissions when spring thaw conditions were simulated (Pelster et al., 2013). Denitrification may be limited in coarse textured soils by C availability when organic matter additions are low, but in cropping systems using GMr, the availability of C and N substrates is likely not a factor.

Most of the previous work on overwinter N_2O emissions from Canadian agricultural soils has demonstrated that the majority of N_2O emissions are a result of denitrification at the soil surface (Furon et al., 2008; Wagner-Riddle et al., 2008; Kariyapperuma et al., 2011). However, another mechanism that contributes to high N_2O emission rates at spring thaw is the physical release of trapped N_2O under a frozen soil layer (Goodroad and Keeney, 1984; Risk et al., 2013). In soils that experience freezing and thawing, water potential and temperature gradients have a significant control on the production, consumption, and location of N_2O within the soil profile (Clough et al., 2005). Therefore, there is a need to understand how soil conditions and cropping

patterns influence the production and movement of N_2O within the soil profile. For a more complete understanding of the soil processes involved in surface gas flux, concentrations of CO_2 and N_2O gas from the soil profile have been studied in a variety of different agricultural (Burton and Beauchamp, 1994; Wagner-Riddle et al., 2008; Yanai et al., 2011; Risk et al., 2014) and forest ecosystems (Billings et al., 1998; Risk et al., 2002; Goldberg et al., 2010; Lavoie et al., 2013), in addition to work done in the Canadian Artic (Brummell et al., 2012; 2014). Risk et al. (2014) reported quantitative evidence of spring thaw related N_2O emissions from the surface being comprised of an initial physical release of N_2O that had accumulated under a frozen soil layer, and a second surface N_2O emission event that was predominately due to N_2O production at the soil surface.

In cold climates, the location, temperature, and amount of soil water has the ability to greatly modify or regulate processes and properties controlling soil gas concentrations and the proportion of N_2O that dissolves in soil water (Heincke and Kaupenjohann, 1999). However, soil freezing will limit water drainage through the soil profile (Kojima et al., 2013) as well as the diffusion of gases out of the soil profile (van Bochove et al., 2001a). Soil ice near the surface may restrict water and gas exchange during winter and allow N_2O to accumulate within the soil profile (Burton and Beauchamp, 1994). As soil ice turns to liquid, supersaturated N_2O in drainage water may contribute to surface N_2O if a pathway for gas exchange with the atmosphere is present (Yanai et al., 2011; Risk et al., 2014) or is transported out of the upper soil profile as drainage water (Burton et al., 2012). Therefore, field management decisions that modify soil water contents and hydrological processes through a change in soil pore size and distribution may also affect N_2O production and release to the atmosphere.

Because N_2O dissolves in water at exponentially higher rates as water temperature drops closer to 0°C (Heincke and Kaupenjohann, 1999) the downward movement of N_2O at spring thaw needs to be considered. The first published report of dissolved N_2O in drainage water was by Dowdell et al. (1979) from tile-drained agricultural fields in South East England. Although this study was published over 35 years ago, this N loss pathway is still not very well understood, and typically not considered in overall N loss budgets. When the amount of N_2O that is stored or newly produced within the soil profile at spring thaw is considered in combination with the amount of water that moves through the soil profile at this time, fields that have improved drainage are potentially emitting a significant amount of N_2O at the drain outlet or drainage

ditches. The effect of a soil's water regime on the concentration and location of dissolved N_2O within a soil profile was described by Heincke and Kaupenjohann (1999) as follows: (i) liquid or solid water acts as a diffusion barrier in soil for N_2O emissions because gas diffusion in air is 10^4 times greater than diffusion in water (Fang and Moncrieff, 1999); (ii) cold soil water acts as a store of N_2O, as N_2O solubility is inversely related to water temperature; (iii) water becomes a rapid mode for N_2O transport through the soil profile when temperature and pressure gradients occur with soil depth.

Agricultural systems that include green manure (GMr) in their crop rotation may emit substantial N_2O emissions that occur outside the growing season, especially in climates that experience extended periods of soil freezing and thawing. Concentrations of CO_2 and N_2O over time and at different soil depths along with details on some soil physical properties provides critical information towards a better understanding of the mechanisms involved during these elevated emissions. The objective of this study was to determine the effects of incorporated red clover on the dynamics of soil profile (7.5–57.5 cm) N_2O concentrations during two soil freezing and thawing seasons (Dec. to Apr.) in Truro, NS. Characterizing and quantifying N_2O losses in water collected from sub-surface drainage tiles and whether they coincide with changes in N_2O content in the soil profile of GMr plots is an additional objective of this study.

4.4 MATERIALS AND METHODS

4.4.1 Field site description and experimental design

All field data were collected at the Brookside experimental site for organic agriculture at Dalhousie University's Agricultural Campus, Truro, Nova Scotia, Canada (45°23'24.72" N lat; –63° 15'16.15" W long). Field data were collected from Dec. to Apr. in Year 1 (2011-12) and 2 (2012-13). Soil and climate at the Brookside experimental site are described in Chapter 3. This study was part of a larger field experiment that included a red clover (*Trifolium pratense* L.) – spring wheat (*Triticum aestivum* L.) crop sequence within a 4-yr rotation (i.e., soybeans–red clover–red clover–spring wheat. The overall experiment was designed in a randomized block design with four main plots (16 x 75 m) and four sub-plots (16 x 16 m) representing all four phases of the crop rotation in each main plot. Each treatment and phase was then replicated over three blocks. The 2nd year of clover (i.e., Year 3 phase of the rotation) was the focus of this study. Treatments differed in management and timing of GMr incorporation using a moldboard plow (Table 4.1). Treatments included: (1) Clover incorporated in Sep. (Early Fall; EF), (2) Clover incorporated in Nov., with aboveground clover biomass removed (Hayed Late Fall; HLF), (3) Clover incorporated in Nov. (Late Fall; LF), and (4) Clover undisturbed until incorporation in May (Spring; SP). A full description of treatments is found in Table 4.1 and the plot layout, including the placement of drainage tiles and gas sampling locations, is included in Appendix 1.

Table 4.1. Description and timing of GMr treatments at the experimental site 2011–2013.

Treatments	Activity	Year 1	Year 2
Hayed Late Fall	Clover biomass harvested as hay	23 June 2011	18 June 2012
Early Fall, Late Fall, and Spring	Clipped clover, residue remained on the soil surface	23 June 2011	18 June 2012
Hayed Late Fall	Clover biomass harvested as hay	18 Aug. 2011	31 July 2012
Early Fall, Late Fall, and Spring	Clipped clover, residue remained on the soil surface	18 Aug. 2011	31 July 2012
Early Fall	Red clover incorporation	17 Sep. 2011	18 Sep. 2012
Hayed Late Fall and Late Fall	Red clover incorporation	2 Nov. 2011	7 Nov. 2012
Spring	Red clover incorporation	2 May 2012	1 May 2013

4.4.2 Climate and soil temperature and moisture monitoring

Soil temperature and volumetric water content were measured in each plot (n=12) using a single 5TE sensor (Decagon Devices, Pullman, WA, USA) at soil depths of 15, 30, and 50 cm. Hourly soil temperatures were logged to an EM-50 data logger (Decagon Devices, Pullman, WA, USA). Sensors were installed by excavating a narrow hole with a tree planter's shovel in each plot, and then the sensor is inserted perpendicular to the surface in the undisturbed sidewall of the narrow hole. Each sensor had a soil volume of influence of 715 cm^3. Air temperature and precipitation were measured on-site using a shielded thermocouple and tipping bucket with hourly measurements logged to a CR200X data logger (Campbell Scientific, Logan, UT, USA).

4.4.3 Drainage water

Sub-surface drainage water was collected through a tile drain system installed to a depth of 0.8 m. Each plot (16 x 75 m) consists of two tile lines placed 6.5 m apart that combined into one line at end of the plot, which drained an area that included all four phases of the rotation. Each line runs into an on-site building with tipping buckets to measure drainage flow (m^3 ha^{-1}). Buffer zones between plots are tile drained as well, hydrologically isolating each plot. Soil water infiltration becomes restricted with soil freezing (Kojima et al., 2013); therefore, periods of limited drainage become an indicator of soil freezing through the soil profile and a rapid increase in drainage indicates soil thawing.

4.4.4 Soil profile N_2O

The concentration and mass of N_2O within the soil profile (7.5–57.5 cm) was evaluated via soil atmosphere sampling tubes, as described by Burton and Beauchamp (1994), Jacinthe and Dick (1996), and Goldberg et al. (2008). Each plot (n=12) had two probes installed down the center, placed 5 m apart by auguring a hole slightly larger than the probe diameter. Probes were installed at least two weeks before the first measurements were made. Soil gas measurements were made from three depths (i.e., 15, 30, 50 cm). Soil atmosphere sampling tubes were installed in 2nd year red clover plots, and were sampled on the same bi-weekly schedule as surface flux measurements beginning in Dec. 2011 until Apr. 2012 (Year 1), and again in Year 2 (Dec. 2012–May 2013).

Total N_2O mass (mg N_2O-N m^{-2}) in each depth (gaseous and aqueous) was calculated using N_2O μL L^{-1} concentrations from each depth and the proportion of the soil pore space that is filled with air (gaseous N_2O) and water (aqueous N_2O). Bulk density measurements from each plot were used to determine the total pore space and measurements of soil water content (vol.) and soil temperature, as described above, were used to determine the proportion of the pore space filled with water or ice.

Calculation of N_2O content in each depth (M_{gas}; mg N_2O-N m^{-3}) was made using;

$$M_{gas} = [N_2O_{gas}]\, V_{gas} \quad [1]$$

where $[N_2O_{gas}]$ is N_2O concentration (mg N_2O-N m^{-3}), V_{gas} is the air-filled porosity (m^3 air m^{-3} soil) estimated in each soil depth using bulk density, and water content at the time of sampling. Hourly soil temperature measurements together with the liquid water content were used to determine the proportion of the soil pore space occupied by soil ice.

The mass of N_2O was determined for the specific volume of air within each soil volume (0.15 m^3) at each sampling depth that had midpoints at 15- (7.5–22.5 cm), 30- (22.5–37.5 cm), and 50-cm (42.5–57.5 cm) and assuming a 1-m^2 soil area. Two bulk density measurements were made in each plot in Nov. of Year 1 and 2 from an intact soil core (0–100 cm) that was divided into four sections (0–15, 15–30, 30–60, and 60–100 cm). Linear interpolation was then used to determine the bulk density for the soil volume associated with gas measurements. The mass of N_2O dissolved in soil water before and during freezing was calculated using Henry's Law as described by Risk et al. (2014).

Henry's law, modified for closed systems, was used to calculate aqueous N_2O content;

$$M_{aq} = M_{gas}\,\beta\,V_{aq}/V_{gas} \quad [2]$$

where M_{aq} is the mass of dissolved N_2O present in each layer (mg m^{-3}), β is the Bunsen absorption coefficient (dimensionless), and V_{aq} is the volumetric liquid water content (m^3 H_2O m^{-3} soil). The Bunsen coefficient used in each calculation was determined as a function of soil temperature at the time of sampling.

4.4.5 Dissolved N_2O in drainage water

Dissolved N_2O in tile drainage water was measured by determining the concentration of N_2O in the pressurized headspace of a sealed glass vial containing water from drainage tiles. During periods of high flow, water samples (4 mL) were taken directly from a spigot at the end of the tile line using a gas-tight syringe and injected into 12 mL glass vials. Prior to sampling, a mercuric chloride ($HgCl_2$) solution (50 µL of 0.02 M) was added to the glass vial to eliminate bacterial activity and the possibility of further gas evolution within the water sample (Roper et al., 2013). A headspace to water ratio of 3:1 has been identified as the optimal ratio for accurate and reproducible results of dissolved gases in water (Jahangir et al., 2012). Background concentrations of dissolved N_2O are assumed to be around 0.25–0.35 µg N_2O-N L^{-1} at 20°C when equilibrium with the atmosphere is reached (Reay et al., 2004a; Minamikawa et al., 2013). Gas analysis took place in Truro, NS, as described in Chapter 3. Aqueous and N_2O gas phase equilibrium is controlled by water temperature and pressure in addition to the solubility of N_2O dissolved in water, which was calculated using a modified method of Weiss and Price (1980) as described by Roper et al. (2013). Corrections to calculated N_2O solubility coefficients for salinity (EC), as described in Weiss and Price (1980), were not necessary for the expected range of drainage water salinity in this study.

4.4.6 Statistical analysis

Soil profile N_2O and CO_2 concentrations were evaluated for the effects of Treatment and Depth over time evaluated using a doubly repeated measures mixed model analysis following $\log_{(10)}$ transformation. Repeated factors were soil Depth and Time for each year and random effects were Block and Block x Treatment. Differences among treatments in N_2O mass within

the soil profile (7.5–57.5 cm) were subjected to a repeated mixed model, which used Block and Block x Treatment interaction term as random factors. Winter (Dec.–Apr.) N_2O emissions and dissolved N_2O in drainage water were evaluated for treatment differences using a repeated measures mixed model that treated Block as the random effect. Treatment means were separated with Tukey's post-hoc test and when a significant interaction was present, means were separated with a test of the simple main effects (slices). Pearson's correlation coefficients were determined to evaluate linear relationships of CO_2 and N_2O using data collected on 12 days from Jan. to Apr. that dissolved N_2O and the soil atmosphere were sampled together over the two year study. All effects were considered significant when $p \leq 0.05$, and statistical tests were performed using JMP software (SAS Institute Inc. Cary, NC, USA).

4.5 RESULTS

4.5.1 Weather, tile drainage, and soil temperature

Air temperatures between years were not different in Feb. or Mar., but were warmer through Dec. in Year 1, and twice as cold through Jan. in Year 2 (Fig. 4.1). Further details on temperature dynamics between years are given in Chapter 3. Total precipitation (rain and snow) in Dec. was similar in Year 1 and 2, while rainfall diminished greatly in Jan. and there were similar snowfalls of 45 and 40 cm, respectively. There was nearly twice as much rain in Feb. of Year 1 as compared to Year 2 (17 mL). Frequent freezing and thawing cycles were recorded at the beginning of Mar. (Fig. 4.1 A, B) as air temperature began to fluctuate ± 10°C around the 0°C mark that initiated a soil thaw. The amount and timing of precipitation during spring thaw had a significant control on the rate of soil thawing. Specifically, during the month of Mar. of Year 1, there was only 15 mm of rain, as compared to 108 mm in Year 2, but snowfall levels during this month were similar, with 47 and 40 cm in Year 1 and 2, respectively. Drainage at spring thaw was nearly five times greater in Year 2, as compared to Year 1 (Fig. 4.1 C, D), which was likely influenced by heavy precipitation events that began in the second week of Mar. as well as greater soil freezing that greatly reduced water drainage prior to the thaw (Chapter 3). This was in contrast to Year 1, when there were very few precipitation events during the month of Mar. and drainage was greater during the period Dec.–Mar. The average water drainage volume from Dec. to the end of Mar. was 948 m^3 H_2O ha^{-1} in Year 1, as compared to 756 m^3 H_2O ha^{-1} in Year 2 when considered across all treatments. The proportion of total drainage for the study period was similar in Dec. between years (i.e., ca. 51%). During Jan., 24 and 12% of the total drainage occurred in Year 1 and 2, respectively. There was minimal drainage through Feb., but 18 and 38% of the total drainage occurred in Mar. of Year 1 and 2, respectively.

The pattern and extent of soil freezing is closely linked to the absence of snow cover and a prolonged period of air temperatures <0°C (Fig. 4.1 E–J). In Year 1, freezing temperatures were immediately followed by temperatures >0°C and this pattern continued for most of the

winter. As a result, soil freezing was minimal in all treatments, as indicated by the increased frequency of water drainage that occurred when soil temperatures were above 0°C. Beginning in mid-Mar., soil temperatures rose quickly in response to a rapid increase in air temperature.

In Year 2, there was an initial period of soil freezing that penetrated to 50 cm and lasted for 1-wk at the end of Jan. (Fig. 4.1). Soil temperatures at all depths quickly rose to 0°C when air temperatures went from –10 to +10°C in a period of 24 h during the first week of Feb. Following this period, air temperatures again remained below 0°C until mid-Feb. when they began to fluctuate above and below 0°C. At the same time, there were periods of rain without any water drainage; an indication that ice had likely formed at the soil surface. What was common between years was the prolonged period that soil temperatures hovered at or very close to 0°C. As discussed in detail in Chapter 3, the Nov. tillage treatments had greater soil freezing than the undisturbed treatment (SP) in Year 2, but no clear differences were observed in Year 1. This effect could be seen at all soil depths in Year 2, but minimal soil freezing at depth in Year 1 was likely why this effect was not observed.

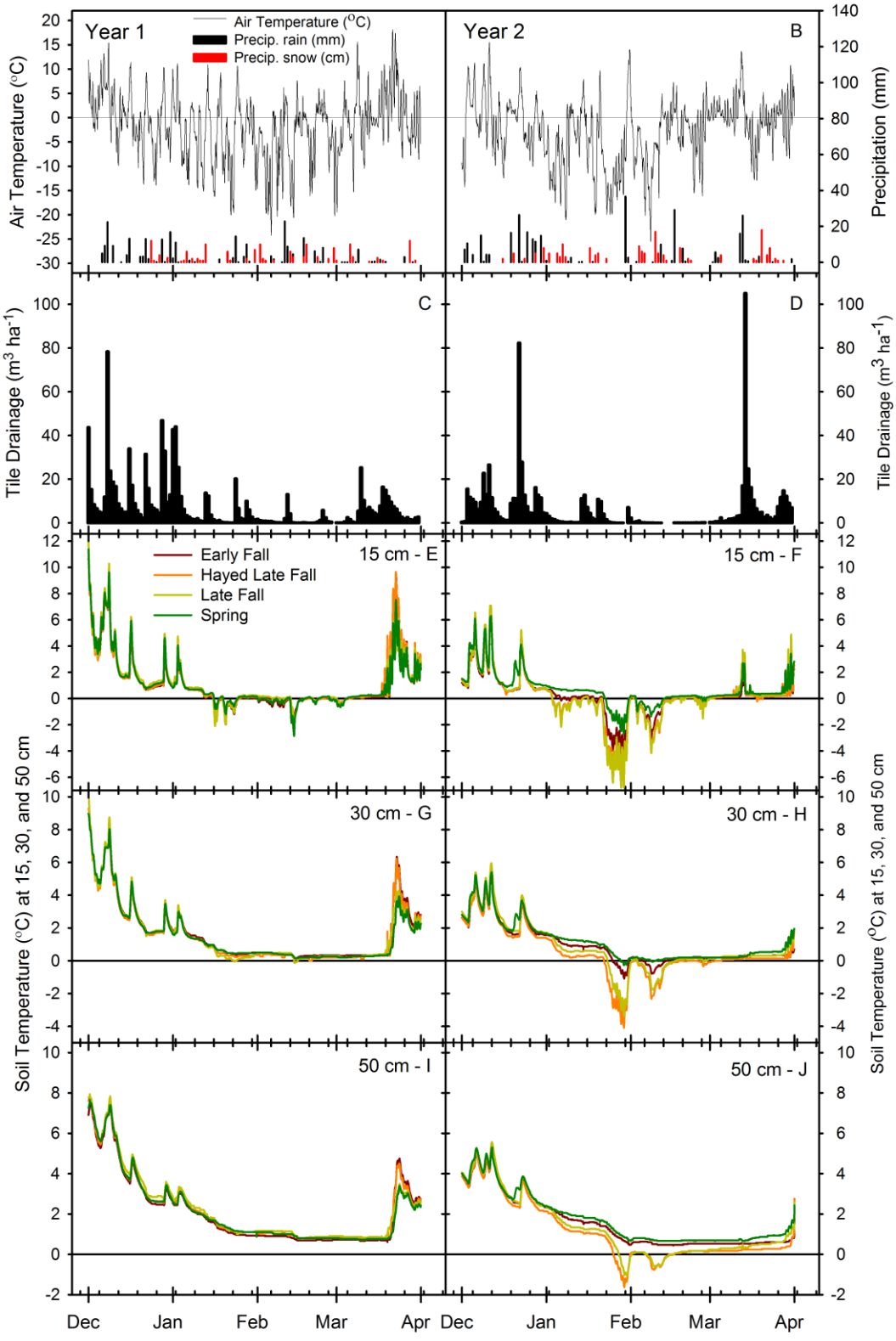

Fig. 4.1. Air temperature and precipitation (snow + rain) measured at the experimental site from Dec. to Apr. in Year 1 (A) and 2 (B). Average drainage water from tile lines (n=12) is shown in panels C and D. Soil temperatures (*n*=3) for clover incorporation treatments from the 15 (E, F), 30 (G, H), and 50 (I, J) cm soil depths.

4.5.2 Soil N_2O concentration

In Year 1, the effect of treatment was significant (p=0.04) when considered across all sample dates and depths (Fig. 4.2). Early Fall and SP tillage treatments had an overall mean N_2O concentration of 36.3 µL L^{-1}, which was much greater than HLF (7.7 µL L^{-1}) while LF was intermediate (15.7 µL L^{-1}). The lowest (p<0.001) N_2O concentrations across all depths and treatments were measured from 1 Dec. 2011 to 17 Jan. 2012 (1.31 µL L^{-1}), while the highest were on 8 Mar. 2012 (65.6 µL L^{-1}). Concentrations declined by 64% on 3 Apr. 2012. There were no differences (p=0.16) in N_2O concentrations among soil depths, nor was there an interaction of Treatment x Depth (p=0.24). There was a Time x Depth interaction (p<0.001) in Year 1 as a result of higher N_2O concentrations measured at lower depths near the end of the study period (22 Mar. 2012 to 3 Apr. 2012).

There were no overall differences (p=0.14) in soil profile N_2O concentrations among treatments in Year 2 (Fig. 4.2), despite a two-fold increase in EF, HLF, and LF treatments as compared to SP (10.6 µL L^{-1}). The highest N_2O concentrations across all treatments and depths were measured at the height of spring thaw (88.4 µL L^{-1}; 13 Mar. 2013); these were quickly followed by the lowest only five days later (18 Mar. 2013), representing an overall change in concentration of 76.88 µL L^{-1}. The Treatment x Time interaction was highly significant (p<0.001) because N_2O concentrations across all soil depths were lowest in the SP tillage treatment on 6th and 13th Mar. 2013. This was followed by the highest concentrations being measured from EF plots that were 1.7, 2.2, and 2.4 times greater than all other treatments on the 4th, 10th, and 25th of Apr. The highest concentrations across soil depths (p<0.001) were measured from the 50 cm depth, which were 48% greater than the mean of 18.29 µL L^{-1} over the 15 and 30 cm soil depths. Similar to Year 1, the Depth x Time interaction was significant (p<0.001), when surface (15 cm) N_2O concentrations were greater than the 30 cm soil depth from 28 Feb. to 6 Mar. 2013. Also similar to Year 1, on 18 Mar. 2013, N_2O concentrations became higher at the lowest soil horizon, which was a trend that continued until the end of the study period on 25 Apr. 2013.

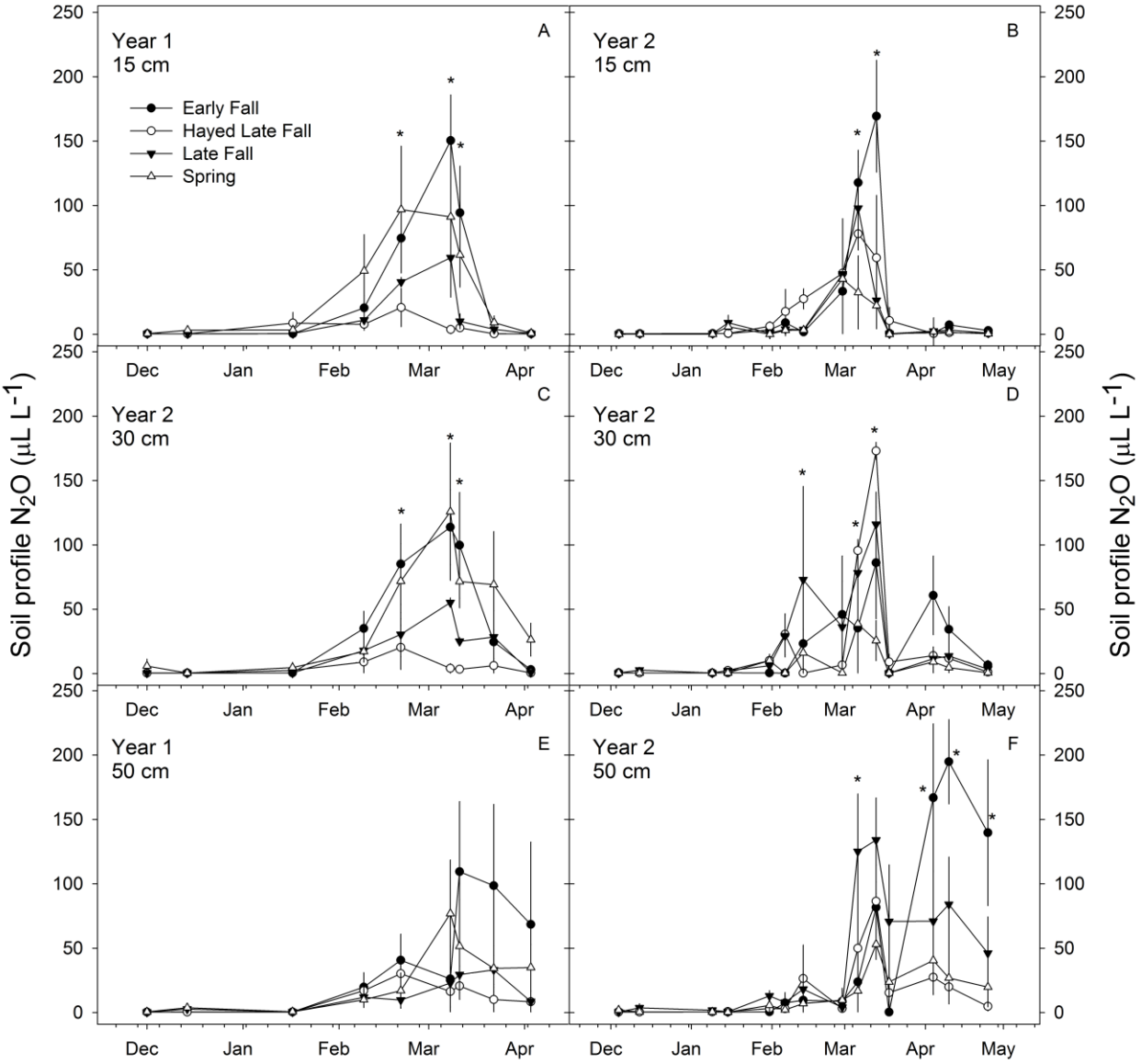

Fig. 4.2. Soil profile N_2O concentrations at 15 (A, B), 30 (C, D), 50 (E, F) cm from Dec. to Apr. in Year 1 and Dec. to May in Year 2. Following a significant Treatment x Time interaction ($p \leq 0.05$) differences among treatments among sample dates are indicated with an asterisk. Error bars represent the SEM (n=3).

4.5.3 Soil CO_2 concentration

In Year 1, the HLF and SP treatments had the highest CO_2 concentrations (12.2 mL L^{-1}) across all soil depths and sampling dates (p= 0.03); these were twice as great as EF and LF (6.2 mL L^{-1}) treatments (Fig. 4.3). Across all depths and treatments, the highest CO_2 concentrations were measured on 8 Mar. 2012 (13.2 mL L^{-1}), as compared to the lowest CO_2 concentrations that were measured on the first sample date in Dec. (4.3 mL L^{-1}). Overall, the greatest (p<0.001) CO_2

concentrations were measured at 50 cm (13.2 mL L^{-1}), which was 56% larger than the mean at 30 cm as compared to the smallest concentrations measured at 15 cm (5.82 mL L^{-1}). A significant Treatment interaction with Depth (p<0.001) was due to the HLF treatment being different than all other treatments (p=0.02) at 50 cm. The Time x Treatment interaction was not significant (p=0.08), but Time x Depth was (p<0.001). The highest (p<0.001) concentrations at 15 cm across treatment were measured between 21 Feb. and 8 Mar. 2012 (11.52 mL L^{-1}), but were reduced by 42% three days later. At 30 cm, the highest concentrations (p=0.01) were measured from 21 Feb. to 11 Mar., until a 51% reduction occurred on 22 Mar. 2012. At 50 cm, the highest (p<0.001) of all concentrations within the soil profile were measured on the last sampling date (4 Apr. 2012), with an average of 22.2 mL L^{-1} across all treatments.

In Year 2 (Fig. 4.3), there was no overall effect of Treatment (p=0.46), but the highest (p<0.001) CO_2 concentrations during the study period were measured between 6 Mar. and 13 Mar. 2013 (10.2 mL L^{-1}). Five days later one of the smallest concentrations (2.5 mL L^{-1}) was measured from the soil profile. On 6 Mar. 2013, SP and EF had CO_2 concentrations that were 2.4 times smaller (p=0.007) than both HLF and LF treatments (13.2 mL L^{-1}), when considered across all soil depths. Across all sample dates, the highest (p<0.001) CO_2 concentrations were measured at 50 cm (7.1 mL L^{-1}), which was more than twice as great as concentrations at 15 cm and 43% more than those at 30 cm. The significant Treatment x Depth interaction (p<0.001) was as a result of the LF treatment having the highest (p=0.01) concentrations on 6 Mar. 2013 at 15 cm, yet the highest concentrations at 30 cm were measured from HLF treatment on this day. Concentrations at 15 cm were at their highest (p<0.001) on 6 Mar. 2013 (12.0 mL L^{-1}) after increasing by 100% in only one week. The next highest CO_2 concentrations were on 13 Mar. 2013 (9.1 mL L^{-1}), which quickly became the lowest concentrations (1.1 mL L^{-1}) only five days later on 18 Mar. 2013. CO_2 concentrations on 13 Mar. 2013 were twice as great as any other date at 30 cm. The lowest CO_2 concentrations at 50 cm were measured during the month of Feb. and the highest during the first half of Apr.

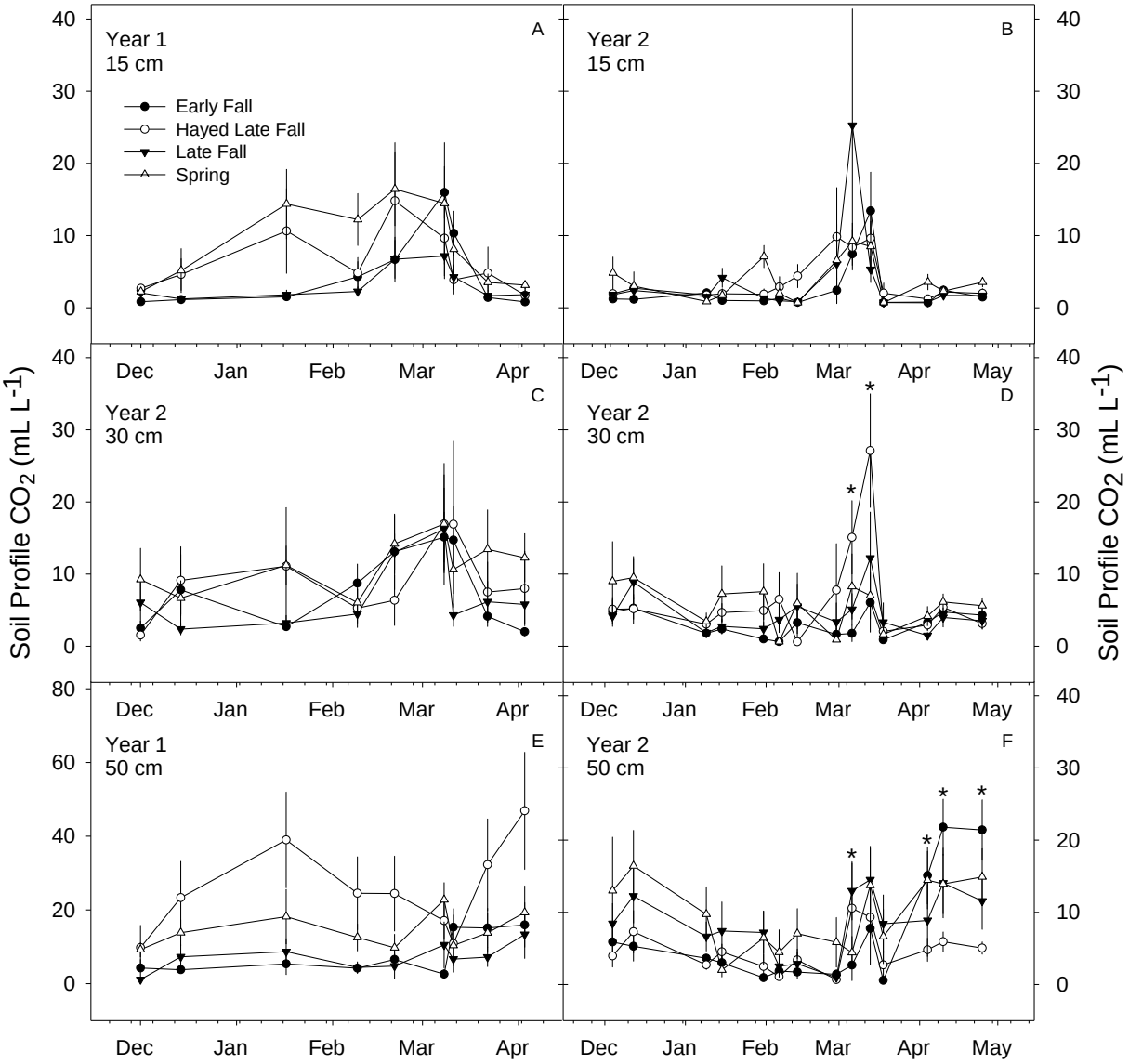

Fig. 4.3. Soil profile CO_2 concentrations at 15 (A, B), 30 (C, D), 50 (E, F) cm from Dec. to Apr. in Year 1 and Dec. to May in Year 2. Take note of the different scale on the Y-axis in panel E. Following a significant Treatment x Time interaction ($p \leq 0.05$) differences among treatments among sample dates are indicated with an asterisk. Error bars represent the SEM (n=3).

4.5.4 Soil profile N_2O content

There was no overall difference in N_2O content (mg N_2O-N m^{-2}) when all depths are summed together among treatments when considered across all sample dates (p=0.09) in Year 1 (Fig. 4.4 A). Across all treatments, the highest (p<0.001) N_2O content was measured on Mar. 6 2012, as compared to the lowest contents measured in Dec., Jan. and Apr. The Treatment x Time

interaction was significant (p=0.001) when the highest N_2O content on 6 Mar. 2012 was measured in both the EF and SP treatments. On 11 Mar. 2012, the EF treatment had the highest N_2O content (37.62 mg N_2O-N m^{-2}) that was 41, 91, 80% greater than SP, Hayed, and LF treatments, respectively.

Across all sample dates in Year 2, there was no overall difference (p=0.21) in N_2O content (mg N_2O-N m^{-2}) among treatments (Fig. 4.4 B). Differences in N_2O content among treatments became evident on 6 Mar. 2013, when Hayed and LF treatments were 92% greater than both the EF and SP treatments that had a mean N_2O content of 15.44 mg N_2O-N m^{-2}. The highest N_2O content across all treatments was measured on 13 Mar. 2013, with a mean of 30.8 mg N_2O-N m^{-2}. The SP treatment had the lowest N_2O content on 13 Mar. 2013 when compared to all other treatments that had at least a two-fold increase over SP. Five days later, total N_2O content decreased to 3.7 mg N_2O-N m^{-2} and was among the lowest of all sample dates. Through the month of Apr., the highest N_2O contents were measured from the EF treatment as compared to all others.

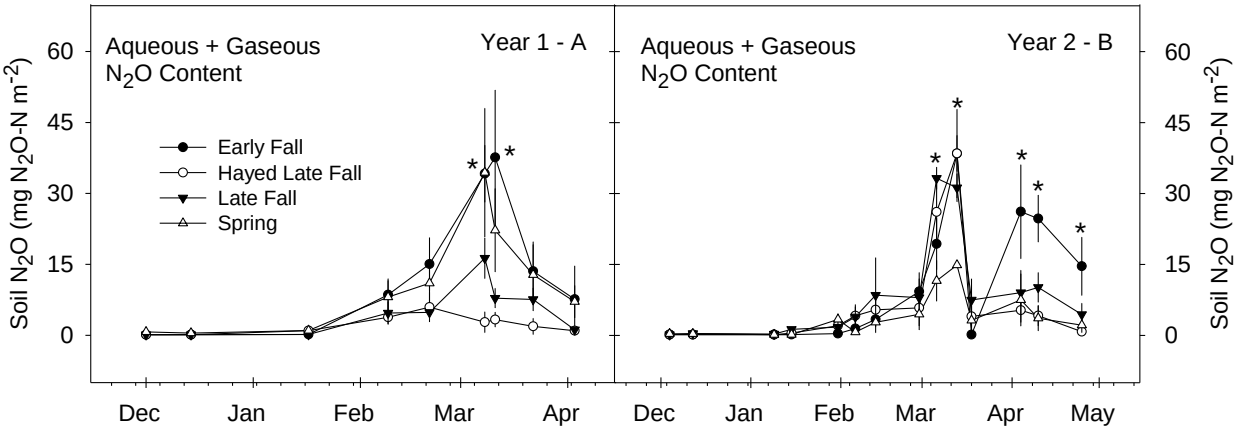

Fig. 4.4. The sum of gaseous and aqueous N_2O content (mg N_2O-N m^{-2}) within the soil profile (7.5–57.5 cm) in Year 1 (A) and Year 2 (B). Following a significant Treatment x Time interaction ($p \leq 0.05$), differences among treatments across sample dates are indicated with an asterisk. Error bars represent the SEM (n=3).

4.5.5 Change in soil N_2O content with surface flux

At the beginning of Feb. in Year 1, N_2O content within the soil profile started to increase (Table 4.1), with the largest increase happening from 21 Feb. to 8 Mar. 2012 in all treatments (18.0 mg N_2O-N m^{-2}). The only exception occurred in the HLF treatment, which decreased by 3.2 mg N_2O-N m^{-2} during this period. On 11 Mar. 2012, the SP and LF treatments had decreases

of 12.2 and 8.5 mg N_2O-N m^{-2}, respectively. This was in contrast to the EF treatment that did not see a decrease in N_2O content until 22 Mar. 2012. With the exception of HLF and SP treatments, N_2O contents (mg N_2O m^{-2}) were comparable to N_2O emissions measured from the soil surface. Surface N_2O emissions measured from Jan. to Apr. from the HLF treatment were an order of magnitude larger than the amount of N_2O measured within the soil profile. Furthermore, SP tillage had N_2O emissions that were nearly twice as large as the amount of N_2O that had accumulated within the soil profile.

Table 4.2. The change in soil profile (7.5 to 57.5 cm) N_2O mass (mg N_2O-N m^{-2}) between sample dates in Year 1.

	Treatments			
Date	Early Fall	Hayed Late Fall	Late Fall	Spring
	- - - - - - - - - - - - - - - - - - mg N_2O-N m^{-2} - - - - - - - - - - - -
01-Dec-11	0.13†	0.11	0.12	0.71
14-Dec-11	0.01	0.01	0.01	–0.23
17-Jan-12	0.03	0.86	0.05	0.54
09-Feb-12	8.41	2.82	4.52	7.05
21-Feb-12	6.48	2.14	0.16	2.93
08-Mar-12	19.11	–3.17	11.45	23.29
11-Mar-12	3.45	0.54	–8.45	–12.2
22-Mar-12	–24.09	–1.43	–0.27	–9.4
03-Apr-12	–5.9	–0.93	–6.41	–5.7
Surface N_2O emissions	- - - - - - - - - - - Jan.–Apr. (mg N_2O-N m^{-2}) - - - - - - - - - - -			
	19.25 (4.9)‡ ab§	15.36 (7.9) ab	11.89 (6.0) b	47.48 (14.7) a
p-value	0.04			

† Positive values represent an accumulation, and negative values represent a decrease in N_2O within the soil depth (7.5–57.5 cm) relative to the previous sample date.

‡ Surface N_2O emissions discussed in Chapter 3 are included here for comparison. Values in brackets represent the SEM.

§ Means within rows with the same lower case letters are not significantly different at the $p \leq 0.05$ level of probability.

In Year 2, there was a rapid increase in N_2O at the beginning of Mar. in all treatments with increases over the next two weeks of 10.4, 23.2, 29.2, and 32.7 in the SP, LF, EF, and HLF treatments, respectively (Table 4.2). Between Mar. 13 and 18, 2013, there was a rapid decrease in N_2O content in all treatments with the biggest change happening in EF (38.29 mg N_2O-N m^{-2}), and the smallest in the SP treatment with a decrease of 11.71 mg N_2O-N m^{-2}. In contrast to all other treatments, the EF treatment accumulated 26 mg N_2O-N m^{-2} from 18 Mar. to 4 Apr. 2013, as compared to only 2.4±1.7 mg N_2O-N m^{-2} in all other treatments. The decrease in soil N_2O in

the EF treatment was similar to N_2O emissions measured at the soil surface. Emissions of N_2O in the SP treatment were less than what had accumulated in the soil profile, while surface emissions from Jan. to Apr. were twice as large as those that were measured in the soil profile for both the HLF and LF treatments.

Table 4.3. The change in soil profile (7.5 to 57.5 cm) N_2O mass (mg N_2O-N m^{-2}) between sample dates in Year 2.

	Treatments			
Date	Early Fall	Hayed Late Fall	Late Fall	Spring
	- - - - - - - - - - - - - - - - - - mg N_2O-N m^{-2} - - - - - - - - - - - - - - - -			
04-Dec-12	0.19†	0.13	0.14	0.32
12-Dec-12	–0.02	0.01	0.25	0.01
09-Jan-13	0.01	0.01	–0.1	–0.17
15-Jan-13	0.01	0.32	0.98	0.02
31-Jan-13	0.19	1.57	0.49	3.2
06-Feb-13	1.07	2.11	2.17	–2.68
13-Feb-13	1.94	1.26	4.58	2.07
28-Feb-13	5.9	0.39	–0.52	1.7
06-Mar-13	10.05	20.29	25.26	7.08
13-Mar-13	19.11	12.38	–2.02	3.33
18-Mar-13	–38.29	–34.45	–23.73	–11.71
04-Apr-13	26.00	1.3	1.56	4.34
10-Apr-13	–1.46	–1.14	–1.22	–3.88
25-Apr-13	–10.08	–3.4	–0.82	–1.46
Surface N_2O emissions	- - - - - - - - - - - Jan.–Apr. (mg N_2O-N m^{-2}) - - - - - - - - - - -			
	35.29 (11.7)‡ a§	62.57 (30.0) a	52.76 (21.6) a	3.6 (1.2) b
p-value	< 0.001			

† Positive values represent an accumulation, and negative values represent a decrease in N_2O within the soil depth (7.5–57.5 cm) relative to the previous sample date.
‡ Surface N_2O emissions discussed in Chapter 3 are included here for comparison. Values in brackets represent the SEM.
§ Means within rows with the same lower case letters are not significantly different at the $p \leq 0.05$ level of probability.

4.5.6 Dissolved N_2O in drainage water

Across all treatments, dissolved N_2O concentrations in tile drainage water from Year 1 (Fig. 4.5 A) were different over all sampling dates (p<0.001), but there was no overall effect of treatment (p= 0.7), nor was the interaction significant (p=0.99). The highest concentration was measured on 9 Mar. 2012 with a mean (±SEM) of 167±16 µg N_2O-N L^{-1} across treatments, as compared to the lowest concentrations measured in Jan. and Feb. with a range of 5±0.5 to 55±14 µg N_2O-N L^{-1}. Cumulative dissolved N_2O-N collected from drainage tiles in Year 1 (Fig. 4.5 E)

was not different among treatments (p=0.64) with a range of 22 to 41 g N_2O-N ha^{-1} from 17 Jan. 2012 to 20 Mar. 2012 (64 d). Mean drainage volume across all plots during this time period was 230±139 m^3 H_2O ha^{-1} (Fig. 4.5 C).

In Year 2, dissolved N_2O concentrations (Fig. 4.5 B) were again not different among treatments (p= 0.72), yet were different with time (p<0.001), but without any interaction with treatment (p=0.33). The highest dissolved N_2O concentration was measured on 13 Mar. 2013 (269±30 µg N_2O-N L^{-1}) and was not different than all other concentrations measured in Mar. 2013 that ranged from 202±25 to 240±34 µg N_2O-N L^{-1}. The lowest concentrations were measured during the month of Jan. 2013 with a mean of 32±10mg N_2O-N L^{-1}. Cumulative N_2O losses in drainage water (Fig. 4.5 F) were again not different among treatments (p= 0.18) with a range of 43 to 115 g N_2O-N ha^{-1} from 19 Jan. 2013 to 23 Mar. 2013 (64 d). Overall, drainage volume across all plots was 266±84 m^3 H_2O ha^{-1} (Fig. 4.5 D).

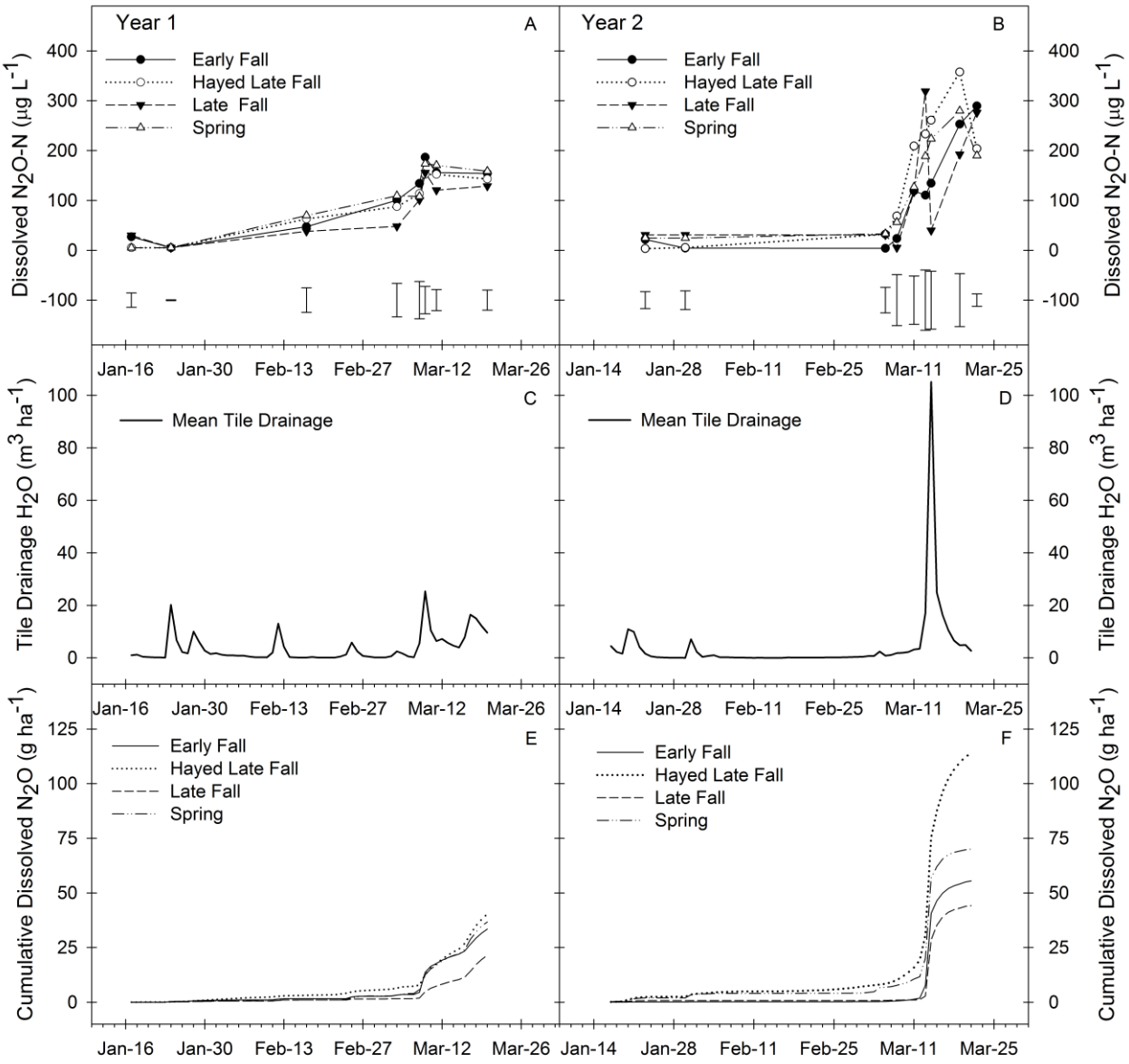

Fig. 4.5. Dissolved N_2O-N (µg L^{-1}) in drainage water (A, B), mean drainage water volume (m^3 ha^{-1}) from all plots (C, D), and cumulative dissolved N_2O-N (g ha^{-1}) (E, F). Error bars in (A, B) represent the average SEM for all treatments (n=3).

4.5.7 Pearson correlation coefficients of soil parameters

Linear relationships of CO_2 and N_2O soil concentrations at multiple depths, daily N_2O and CO_2 emissions, and dissolved N_2O concentrations in drainage water on six dates between Jan. and Apr. were selected in each year and were evaluated using Pearson's correlation coefficients (Table 4.3). As observed in Chapter 3, emissions of CO_2 and N_2O at the soil surface were highly correlated (r=0.81; p<0.001; n=46). Soil concentrations of CO_2 and N_2O in space and over time were positively related at 30- (r=0.76; p<0.001; n=46) and 15-cm (r=0.70; p<0.001; n=46) as well. Temporal variations in N_2O concentrations at 30 cm were closely related to both 15- (r=0.72; p<0.001; n=46) and 50-cm (r=0.65; p<0.001; n=46), while CO_2 at 30 cm was most related to CO_2 at 15 (r=0.49; p<0.001; n=46) when compared to 50 cm (r=0.30; p<0.04; n=46).

The size of the daily N_2O emission event had strong positive relationships with N_2O concentrations at 30- (r=0.52; p<0.001; n=46) and 50-cm (r=0.45; p=0.001; n=46), but no relationship was found for the N_2O concentration at 15 cm (r=0.01; p=0.99; n=46). Interestingly, there were no significant relationships of daily CO_2 emissions with 15 cm CO_2 or N_2O concentrations either, yet Soil CO_2 concentrations at 30 cm were positively related to surface N_2O flux (r=0.42; p=0.003; n=46). Another relationship indicating the temporal relationships of winter N loss is the increase in surface flux events with the concentration of N_2O dissolved in drainage water (r=0.44; p=0.002; n=46). Furthermore, soil N_2O concentrations were positively related to dissolved N_2O concentrations collected in drainage water for N_2O at soil depths of 50- (r=0.58; p<0.001; n=46) and 30-cm (r=0.43; p=0.002; n=46).

Table 4.4. Pearson correlation coefficients (*r*) among soil gas concentrations and N_2O losses to air and drainage water on 12 sample days from Jan. 1 to Apr. 1 in Years 1 and 2.

	Daily N_2O surface flux†	Daily CO_2 surface flux	Soil gas					
			$[CO_2]$ 15 cm	$[N_2O]$ 15 cm	$[CO_2]$ 30 cm	$[N_2O]$ 30 cm	$[CO_2]$ 50 cm	$[N_2O]$ 50 cm
Dissolved N_2O (µg N_2O-N L^{-1})	0.44**‡	0.34*	0.04	0.25	0.22	0.43**	–0.06	0.58***
Daily N_2O surface flux		0.81***	–0.002	0.000	0.42**	0.52***	0.05	0.45***
Daily CO_2 surface flux			–0.15	–0.15	0.24	0.27	0.04	0.23
Soil $[CO_2]$ 15 cm				0.70***	0.49***	0.52***	0.37**	0.27
Soil $[N_2O]$ 15 cm					0.47***	0.72***	–0.04	0.34*
Soil $[CO_2]$ 30 cm						0.76***	0.30*	0.27
Soil $[N_2O]$ 30 cm							0.08	0.65***
Soil $[CO_2]$ 50 cm								0.16

† Dissolved N_2O (µg N_2O-N L^{-1}) refers to drainage water N_2O concentration on the day of sampling. Daily N_2O and CO_2 flux refers to the interpolated values for that day. Soil gas refers to N_2O and CO_2 concentrations within the soil profile.

‡ Six sampling events in each year (Jan.–Apr.) and 4 treatments created a total sample size of 48 (*n*= 46) for the correlation matrix. * Significant at the 0.05, ** 0.01, and *** 0.001 levels of probability.

4.6 DISCUSSION

4.6.1 Soil profile N_2O and CO_2 concentrations and content

Elevated N_2O concentrations within the soil profile during frozen soil conditions are caused by increased production without further reduction to N_2 and the lack of diffusion or mass-flow out of the soil profile. Before an increase in N_2O within the soil profile in this study, there were several rapid thaw and re-freeze events in Jan. and Feb in both years, although these were greater in magnitude in Year 2 (Fig. 4.1). Similar relationships between soil temperature and increasing soil N_2O concentrations in soils that experience freezing and thawing conditions have been observed frequently (Maljanen et al., 2007; Yanai et al., 2011; Risk et al., 2014). Snowmelt or precipitation that freezes at or near the soil surface fill soil pores with ice that effectively seals the soil profile from the atmosphere and has been suggested to be a mechanism that initiates soil N_2O accumulation (Burton and Beauchamp, 1994; Risk et al., 2014). Soil concentrations need to be considered with respect to surface conditions, as frozen soil layers close to the surface might limit the upward diffusion and downward movement of soil water. The highest N_2O concentrations measured within the soil profile in this study (ca., 150 μL L^{-1}) were similar to those measured by Risk et al. (2014) near the soil surface, but our measurements were an order of magnitude larger at 30 and 50 cm as compared to those measured by Risk et al. (2014). However, this trend was not observed in all blocks in our study, as indicated by the large variation during many sampling dates. Burton and Beauchamp (1994) also described a high amount of variability in soil N_2O concentrations; although field variability is anticipated, the magnitude of differences among treatments in our study were somewhat surprising. However, the ratio of NO_3^- to available C together with the aeration status in soil is well known to affect which product of denitrification will be dominant. The timing and season of GMr incorporation would therefore affect the availability of substrate needed during denitrification, which would allow the production of N_2O to vary among treatments. The varied effects of soil freezing

between years and treatments could also help explain the high variance in soil gas concentrations.

Soil N_2O contents before spring thaw differed among treatments in both Year 1 and 2, but the trend among treatments was not consistent. In particular, the undisturbed GMr treatment (SP) had the highest N_2O content in Year 1 and the lowest in Year 2; a trend that was observed for surface emissions as well (Chapter 3). Indeed, the positive relationship of N_2O concentrations at 30 and 50 cm with daily surface flux (Table 4.4) indicates that N_2O dynamics within the soil profile related to similar differences in daily N_2O flux at the soil surface. The very weak relationships between surface flux and both N_2O and CO_2 concentrations at the 15 cm soil depth was surprising, but may reflect the fact that N_2O at 15 cm was already emitted at the soil surface, had experienced a reduction to N_2, or were transported to lower soil depths with moving soil water. When the amount of N_2O within the soil profile is scaled up from m^{-2} to ha^{-1} for a comparison with surface emissions that occurred during the month of Mar. (spring thaw), the relationship between soil accumulation and emission at the soil surface is again inconsistent among treatments in each year. This was true for all treatments except EF that had increased soil profile N_2O content and then reductions that were consistently greater than the measured emissions at the soil surface by 90 and 182 g N_2O-N ha^{-1} in Year 1 and 2, respectively. This pattern was also seen in the SP treatment in Year 2. However, the treatments with the largest spring thaw N_2O emissions in Year 1 and 2 had more N_2O emitted from the soil surface than what had accumulated in the soil profile by a factor of 50% for SP in Year 1 as compared to 36 and 41% for HLF and LF treatments in Year 2. The change in profile N_2O content and the amount emitted from the soil surface during spring thaw was similar for HLF and LF in Year 1 with surface emissions of 41 and 102 g N_2O-N ha^{-1}, respectively. Hydrological controls and substrate availability were different among years and could help explain why there were inconsistencies among treatment effects on soil N_2O content and emissions from the soil surface when compared over two years. Overall, N_2O concentrations within the soil profile were a good indicator of the potential for N_2O emissions at the soil surface; treatments with the greatest surface emissions at spring thaw also had the highest profile N_2O concentrations.

Because concentrations of CO_2 and N_2O in the soil atmosphere are a result of microbially-mediated processes (Amundson and Davidson, 1990), it should not be too much of a surprise that CO_2 and N_2O concentrations were highly correlated at 15 (r = 0.70; p<0.001; n=46)

and 30 cm (r=0.76; p<0.001; n=46) soil depths. Because of the very high concentrations of both of these gases before spring thaw, there is good evidence of limited diffusion from the soil profile to the atmosphere. The similar dynamics of CO_2 and N_2O concentrations within the soil profile as well as emissions from the soil surface over time suggests that soil physical properties have a strong control on the overall potential for surface emissions and the duration of increasing and decreasing soil gases. Meanwhile, the difference among treatments in each year suggests that more of the biological controls and substrate availability will dictate the magnitude of these changes. Cropping systems with high C inputs have been shown to have high N_2O emissions as a result of the localized anaerobic conditions created by the increase in biological activity (Petersen et al., 2013), however it is unclear what the effect would be in cold soils with limited soil gas exchange with air above the soil surface. Furthermore, the expression and activity of N_2O reductase enzyme in cold soils has been identified as being a key process that may lead to N_2O being the dominant product of denitrification (Smith et al., 2010; Wertz et al., 2013; Németh et al., 2014; Tatti et al., 2014).

Changes in surface conditions due to timing of GMr tillage may have other effects on the biological community. Plant cover has the ability to trap more snow as compared to bare soil surfaces, which lessens the effects of air temperature fluctuations and reduces the extent of soil frost. The actions of soil freezing and thawing can break up soil aggregates that increase substrate availability that was otherwise physically protected within the aggregate. Available C and N also increase as a result of plant and soil animal cell lysis, which is known to occur following a freeze thaw–cycle (Cleavitt et al., 2008; Elliott and Henry, 2009). Furthermore, restricted outward diffusion of soil gas, as indicated by high gas concentrations, also implies that the inward diffusion of O_2 is restricted, which would create anaerobic conditions that are necessary for denitrification and N_2O production (Amundson and Davidson, 1990). While the pattern of soil freezing events was similar across treatments, soil temperatures were consistently lower in both the HLF and LF treatments and intermediate for the EF, when compared to SP, at all soil depths in Year 2. This result was surprising, but was likely a factor of the Nov. plowing that occurred in both of these treatments and the influence that plant cover has on the ability to trap snow and insulate the soil from freezing conditions. Air-filled porosity and the size of soil pores also influence the extent and degree of soil freezing (Parkin et al., 2013). Surface conditions and a change in the distribution and size of soil pores would be modified by inversion

tillage and could explain why there was a difference in soil freezing among treatments at all soil depths in Year 2 (Fig. 4.1).

4.6.2 Dissolved N_2O in tile drainage flow

Because GMr tillage treatments only represented 25% of the area that contributed to water collected from drainage tiles, it is not surprising that there were no differences in Treatments in either year of study (Fig. 4.5 E, F). The variability in dissolved N_2O concentrations among treatments (Fig. 4.5 A, B) that was evident in both years is not necessarily due solely to differences in the timing of clover incorporation, but more of an indication of the variable site conditions that influence both N_2O production and the hydrological properties. However, the similar timing for increasing dissolved N_2O concentrations at spring thaw with the decrease in soil N_2O concentrations from GMr plots in both study years indicates that this rarely measured pathway for N_2O losses can be significant. The greatest dissolved N_2O concentrations were measured on the same day as the largest drainage events in both Year 1 and 2. In Year 1, 59% of the total drainage volume from Jan. to Apr. occurred during the month of Mar. as compared to 79% in Year 2. Nitrous oxide contents in drainage water from Year 2 were nearly double those in Year 1 because of greater drainage, and higher overall concentrations during spring thaw.

In Year 2 of the study, we started measuring dissolved N_2O losses earlier in the fall with mean N_2O losses being 44 g N_2O-N ha^{-1} from 16 Oct. to 1 Jan (data not shown). Although there was 5-fold increase in dissolved N_2O in Year 2 when compared to Year 1 when a 184 d sampling period was considered, much of this difference is due to the longer sampling period and an increase in drainage water volume at spring thaw. When the amount of dissolved N_2O in drainage water was calculated on a daily or water volume basis there were no statistical differences among years (data not shown). Dissolved N_2O concentrations were highest at spring thaw, but low concentrations with high amounts of drainage volume during the late fall and early winter will contribute to annual N_2O losses in drainage water. Another point to consider is the fact that N_2O concentrations in drainage water had not declined to their lowest levels at the end of our study periods, and therefore, there is a possibility for more N_2O losses that were unaccounted for in the spring and early summer during this study.

The positive relationship (Table 4.4) of dissolved N_2O concentrations with N_2O concentrations at 30- and 50 cm (r=0.43–58; p<0.01; n=46) and surface N_2O flux (r=0.44; p=0.002; n=46) indicate that at this site the depth and magnitude of N_2O concentrations through time are an effective indicator of the timing and potential for N_2O losses from the soil surface and N_2O dissolved in drainage water. During periods of high drainage and elevated N_2O concentrations within the soil profile, as observed during Mar. in each year at this study site, there was also an increased risk for dissolved N_2O in drainage water. It is unclear if effective and rapid water drainage from the installation of tile drainage has the potential to reduce surface N_2O flux from the soil surface, or merely transfers the emissions to the drainage ditch or outlet. Further investigation of the relationship of improved drainage and related spring thaw N_2O losses is needed.

Most of the studies assessing dissolved N_2O losses from agricultural areas have been focused during the growing season, or do not have extended freezing and thawing periods in which high concentrations of N_2O can accumulate under a restricted layer. Heincke and Kaupenjohann (1999) reviewed N_2O-N concentrations in drainage water that had a range from 25 to 132 µg N_2O-N L^{-1}. Reay et al. (2004a) reported comparatively smaller N_2O-N concentrations in drainage water (2–4 µg N_2O-N L^{-1}) that resulted in an annual loss of 107 g N_2O-N ha^{-1} yr^{-1}, and indirect N_2O emissions via drainage water were considered to be insignificant in Scottish agricultural systems. From Jan. to Apr. in our study, there was a mean of 33±13 and 70±34 g N_2O-N ha^{-1} lost as dissolved N_2O across all treatments in Year 1 and 2, respectively, which was 14 and 18% of N_2O-N emitted from the soil surface during the same period (Tables 4.1 and 4.2). Reay et al. (2004b) measured dissolved N_2O losses to be <1% of the N_2O emitted at the soil surface over 30 days following spring fertilization of a barley field in the U.K, but likely did not have the same increase in soil N_2O concentrations that were measured here prior to spring thaw. A lysimeter study from an intensively cropped volcanic ash soil in Japan reported annual N_2O losses in drainage water of 114 g N_2O-N ha^{-1} yr^{-1}, which was in comparison nearly half of the measured total emissions from the soil surface (Minamikawa et al., 2013). Therefore, N_2O losses from drainage water can be considerable in some systems, especially when considered over the entire year. Additionally, diffusion restrictions imposed by frozen soil layers, as seen in this study, increases the likelihood of rapid N_2O losses in drainage water. Finally, cumulative dissolved N_2O and maximum daily flux losses (g N_2O-N ha^{-1}) among

treatments were similar to each other, and were occurring concurrently at the peak of the spring thaw drainage event.

4.7 Conclusions

Nitrous oxide dissolved in drainage water represents a N_2O loss pathway that could lead to increased N_2O emissions to the atmosphere, especially when fields have drainage tiles installed. There was a mean of 33 and 70 g N_2O-N ha^{-1} measured in drainage water in Year 1 and 2, respectively, which represents a loss of N_2O from these fields that were comparable to the highest daily emission events that occurred at spring thaw. The buildup of N_2O within the soil profile was similar in magnitude between years with a maximum of around 40 mg N_2O-N m^{-2} in both years. The disappearance of N_2O within the soil profile was rapid in both years and occurred at the peak of the spring thaw drainage, but then climbed again when drainage slowed. Significant correlations indicated that the disappearance of N_2O from within the soil profile was also closely timed with N_2O emissions at the soil surface and the increase in dissolved N_2O concentrations in drainage water, yet GMr incorporation treatments had varying effects on N_2O production within the soil profile and subsequent loses to the environment from year to year. However, the change in N_2O content from within the soil profile occurred over a shorter period of time in Year 2 as compared to Year 1 due to the increase in drainage in Year 2, and the accumulation of N_2O within the soil profile that occurred later in winter. The highest N_2O contents among treatments also coincided with the highest emissions from the surface at spring thaw; an indication that the concentration of N_2O within the soil profile is also good predictor of N_2O production, and the potential for emissions from the soil surface or being emitted in drainage water during spring thaw.

5. SEASONAL VARIATION HAS A GREATER INFLUENCE ON SOIL N AVAILABILITY AND CROP N UPTAKE THAN THE TIMING OF GREEN MANURE TILLAGE IN ORGANIC WHEAT PRODUCTION

5.1 Preface

One of the greatest challenges for agricultural production systems that use green manures (GMr) for soil fertility is to manage the rate and amount of NO_3^- production to match requirements of the following cash crop. Soil N supply (SNS) becomes unsynchronized with plant uptake, due to atypical spring soil conditions or crop failure, which will result in elevated soil NO_3^-concentrations that then become susceptible to N losses, such as NO_3^- leaching and denitrification, especially in humid regions in Atlantic Canada. Emissions and dynamics of N_2O from the GMr phase of the rotation during the off-season were the foci of the previous research chapters, whereas in this chapter the focus is on the agronomic performance of the cash crop (wheat), SNS, associated in-season N_2O-N losses, plus annual NO_3^- leaching. I evaluated how fertility management systems (FMS) affected wheat N uptake and yield, with respect to in-season N_2O emissions. Two treatments are different in this chapter because of the addition of fertilizer in the EF treatment and manure application in HLF before wheat planting. Due to the nature of the experimental design, each phase of the 4-yr rotation was present in every year; thus, annual loss and monthly mean NO_3^- concentrations from subsurface drainage tiles are interpreted for the entire 4-yr crop rotation.

5.2 Abstract

A 2-yr study was conducted in Truro, NS, Canada to investigate potential fertility management systems (FMS) that provide the highest spring wheat yields with minimal NO_3^- leaching, and in-season N_2O emissions. Four FMS differed in the timing of green manure (GMr) tillage and the use of supplemental N before spring wheat planting as follows: (1): Sep. incorporated GMr and spring application of 70 kg NH_4NO_3 ha^{-1} (Early Fall + N_{70}; EF+N_{70}); (2) Nov. incorporated GMr with GMr hay removed and spring application of manure at 70 kg N ha^{-1} (Hayed Late Fall + M_{70}; HLF+M_{70}); (3) Nov. incorporated GMr with no supplemental N (Late Fall; LF); (4) May incorporated GMr with no supplemental N (Spring; SP). There were no differences in wheat N uptake or grain yield among FMS in either year of study. However, there was a 76% increase (94.5 ± 19.9 to 166 ± 7.5 kg N ha^{-1}) in N uptake and wheat grain yield that was 2.5 times greater (+1583 kg ha^{-1}) in Year 2 than Year 1. Differences in N uptake between years were also supported by more soil available N_{min} in Year 2 and a 44% decrease in annual NO_3^- leaching (15.5 kg NO_3^--N ha^{-1}), which was an indication of a soil N supply (SNS) that was better synchronized for plant uptake in Year 2. The 2-yr mean NO_3^--N concentration in drainage water was greatest from the EF+N_{70} (10.7 mg NO_3^--N L^{-1}) FMS during the fall and early spring, as compared to the other FMS that had a mean value of 7.7 mg NO_3^--N L^{-1}. In-season N_2O emissions from the wheat crop did not differ among FMS, with the exception of an increase in N_2O on two sample dates following manure application in the HLF+M_{70} FMS in Year 1, but were otherwise low (0.09–0.87 kg N_2O-N ha^{-1}). Weather variation between years had a greater control on in-season SNS and overall N dynamics than FMS alone. There was no agronomic advantage of using supplemental N, especially when the additional costs for application and the potential for additional N losses are considered.

5.3 Introduction

The balance between sufficient N availability for plant growth while minimizing environmental N losses is a challenge in all agricultural systems, especially those that are managed organically. There have been numerous examples of reduced N losses in organic production systems that rely on organic N sources, as compared to conventional systems that rely

on mineral fertilizers for soil fertility (Drinkwater et al., 1998; Stopes et al., 2002; Eriksen et al., 2004). Furthermore, systems that rely on biologically fixed N as the main N input are characterized as having lower whole-farm GHG emissions per hectare (Drinkwater et al., 1998; Olesen et al., 2006), better soil C storage and sequestration (Hepperly et al., 2007; Zikeli et al., 2013), soil health (Lynch, 2014), and lower whole-farm energy use (Refsgaard et al., 1998; Lynch, 2009; Lynch et al., 2011). In contrast, yield-scaled N_2O emissions are higher in organic systems when compared to conventional management (Skinner et al., 2014), and NO_3^- leaching from red clover GMr can be higher than systems that rely only on mineral fertilizer application (Bergstrom and Kirchmann, 2004).

Total N inputs from GMr and animal manures used in organic management have been shown to be comparable to conventional systems (Berry et al., 2002), yet yields can be limited by N availability (Lynch et al., 2012). Nitrogen losses and the yield of organically managed cash crop rotations can be dramatically influenced by in-season soil N supply (SNS) (Koraseth et al., 2002), which is described as the sum of available N from the mineralization of soil organic matter, crop residues, and organic amendments in addition to any residual N from the previous growing season (Dessureault-Rompre et al., 2015). Because of year to year variation in soil temperature and water content, the timing of the SNS is difficult to predict (Griffin, 2008; Dessureault-Rompre et al., 2012). As a result, the timing of the SNS from GMr to the subsequent crop can become unsynchronized with crop uptake and results in lower yields, rather than being a result of low total N input (Berry et al., 2002). Management choices for GMr that could increase soil N uptake by the cash crop involve adjusting the timing and season of incorporation, growing for forage, or using the biomass for anaerobic digestion before applying it to the field (Lynch et al., 2012; Brozyna et al., 2013).

The transfer of N from GMr to a subsequent cash crop varies widely due to the legume species grown and the climate and soil type being studied. However, N inputs that are in the range of 150 to 300 kg N ha^{-1} are possible (Berry et al., 2003). Potatoes in eastern Canada were supplied with 100 to 125 kg N ha^{-1} following a legume (i.e., red clover or vetch) GMr (Lynch et al., 2012), which was comparable to a crop uptake of 99 kg N ha^{-1} from potatoes supplied with N by an alfalfa/timothy plowdown in northeastern USA (Sharifi et al., 2008). Spring incorporation of GMr better protects against N losses outside of the growing season, but spring SNS for N demanding crops may not be well synchronized if soil mineralization and nitrification processes

are affected by soil conditions that either slow, or increase their rates prior to the planting of the cash crop. Furthermore, the timing of spring tillage can be problematic due to wet soil conditions that may limit field accessibility and delay the planting of the successive crop. In Atlantic Canada, SNS and the timing of N uptake by the successive crop may have to be managed by incorporating the GMr in the fall, or by including supplemental nitrogen sources in the spring before planting (i.e., fertilizer or manure) because of the shorter growing season.

Leaching of NO_3^- is controlled by N mineralization rates, crop uptake, manure or fertilizer additions, climate and soil types, and the timing of tillage (Dinnes et al., 2002). For example, given the combination of coarse textured soils and high fall precipitation that is characteristic of areas with Podzolic soils, the majority of the SNS comes directly from in-season mineralization (Zebarth et al., 2009). Agricultural systems in these types of soils and climate therefore have a greater risk of N leaching (De Jong et al., 2007; Zebarth et al., 2009), not only from high N application rates, but low crop uptake as well (Zebarth et al., 2009). However, there can be considerable NO_3^- production in the fall following legume residue incorporation (Sanderson and MacLeod, 1994; Dahlin et al., 2005), which also becomes susceptible to leaching. November soil profile (0–90 cm) NO_3^--N content from a sandy loam Podzol in Atlantic Canada was greatest when red clover was incorporated with a moldboard plow in Sep. (82.7 kg N ha^{-1}), as compared to Oct. incorporation (41.0 kg N ha^{-1}), or was left undisturbed (20.3 kg N ha^{-1}) through the winter (Sanderson et al., 1999). The greatest NO_3^- leaching observed in a 6-yr crop rotation from Denmark, with varied animal stocking densities and manures, occurred when a legume/grass phase of the rotation was incorporated (Eriksen et al., 2004). Similarly, 20% less total N was leached in autumn when NH_4NO_3 was spring applied rather than the spring incorporation of GMr in Sweden (Bergstrom and Kirchmann, 2004). The season and timing of GMr incorporation needs to be considered when the reduction of NO_3^- leaching is an objective of the agricultural system.

Similar to NO_3^- leaching, field management decisions that increase N use efficiency (NUE) and reduce residual soil mineral nitrogen (RSMN) in cropping systems are also critical for effective N_2O emission mitigation (Smith et al., 2008; Van Groenigen et al., 2010). Denitrification is the likely N_2O producing pathway following animal manure slurry application or legume GMr incorporation, but specific soil conditions and inputs of C and N will determine the proportion of N_2O produced during denitrification (i.e., molar ratio; $N_2O:N_2+N_2O$). Drury et

al. (2008) reported the greatest field N_2O emissions were from monoculture corn (2.62 kg N ha^{-1}) as compared to 1.03 kg N ha^{-1} from a soybean–corn–wheat rotation. Large differences in N losses between the two systems were attributed to the high rates of fertilizer use, and the size of the RSMN pool in continuous corn. In contrast, Rochette et al. (2004) determined that soils fertilized with liquid manure had greater emissions over a short period of time, as compared to those systems that rely solely on mineral fertilizers for fertility. The triggers for these emissions were the high availability of NH_4^+ and labile C in manure slurry coupled with precipitation events that increased water filled pore space (WFPS) in soil to levels that favored denitrification with a high N_2O:N_2+N_2O ratio (Chantigny et al., 1998).

Reducing environmental N loss is an important environmental goal for agricultural production systems; in addition, it can improve the economic performance of operations by reducing the cost of purchased N inputs in the form of inorganic fertilizers or manure (Roberts et al., 2008; Zebarth et al., 2009; Novak and Fiorelli, 2010). Legumes grown as GMr have the ability to supply sufficient N to successive crops, but there is a risk for N losses if the SNS becomes unsynchronized with crop demand (Koraseth et al., 2002). The objective of this study was to determine how four FMS using red clover as green manure for spring wheat production in Atlantic Canada affected spring wheat grain yield and N uptake along with in-season N_2O emissions and annual NO_3^- leaching from subsurface drainage tiles.

5.4 MATERIALS AND METHODS

5.4.1 Field site description and experimental design

All field data were collected at the Brookside experimental site for organic agriculture at Dalhousie University's Agricultural Campus, Truro, Nova Scotia, Canada (45°23'25.2918; –63°15'21.1932). Soil at the Brookside experimental site as well as a climate description was provided in Chapter 3. Drainage at the site was further improved by the installation of subsurface drainage tiles at a depth of 0.8 m. A diagram of tile lines and the plot layout is found in Appendix 1.

This study was part of a larger project that was looking at year round N dynamics and losses to air and water within organic grain rotations. All field data were collected from June 1st to Sep. 10th over two years (2011 and 2012) from plots planted to spring wheat (*Triticum aestivum* L.) that followed red clover (*Trifolium pratense* L.) grown as GMr for two years. Crop rotation consisted of soybeans (year 1), two years of red clover (years 2, 3), and spring wheat (year 4) as the cash crop. The experiment was designed as a randomized block design with four main plots (fertility management treatments; 16 x 75 m) replicated over three blocks. Each main plot (n=12) consisted of four sub-plots (crop rotation phase; 16 x 16 m) that represented all four phases of the rotation during each year of study. Each main plot (16 x 75 m) had two drainage lines placed 6.5 m apart that combined into one line at end of the plot. Between each plot was a tile-drained buffer zone (4 m) that hydrologically isolated drainage water from each plot before being diverted into an on-site heated building with tipping buckets installed below the frost line that measured drainage flow and allowed for year-round water sampling.

Treatments (i.e., FMS) differed in timing of red clover incorporation and the use of supplemental N (i.e., inorganic fertilizer and liquid dairy manure) applied prior to the spring wheat phase of the rotation. Clover was incorporated with a moldboard plow to a depth of 20 cm. Fertility Management Systems were: (1) Clover incorporated in Sep. + NH_4NO_3 fertilizer applied at 70 kg N ha^{-1} prior to wheat planting (Early Fall + N_{70}; EF+N_{70}), (2) Clover incorporated in

Nov., but aboveground clover biomass was removed as a hay crop and liquid dairy manure applied at 70 kg total N ha^{-1} in May prior to wheat planting (Hayed Late Fall + M_{70}; HLF+M_{70}), (3) Clover incorporated in Nov. (Late Fall; LF), and (4) Clover incorporated in May (Spring; SP).

In this study, we focused on in-season N dynamics during the spring wheat phase (Year 4) of the rotation and annual NO_3^- leaching from all four phases of the rotation. The wheat variety used in this study (AC Helena) was described by Nass et al. (2001) and weed control was accomplished using standard practices for organic grain management (i.e., heavy seeding rate, 190 kg ha^{-1}; and post-emergent weeding using a flex-tine harrow) as described by Shirtliffe and Benaragama (2014). The timing of all field activities and treatment applications is presented in Table 5.1.

Table 5.1. Description and chronological timing of field operations among Fertility Management Systems (FMS) during the two years of study.

Treatments (FMS)	**Activity**	**Year 1**	**Year 2**
Early Fall + N_{70} (EF+N_{70})	Red clover incorporation	14 Sep. 2010	17 Sep. 2011
Hayed Late Fall + M_{70} (HLF+M_{70})	Red clover incorporation	2 Nov. 2010	3 Nov. 2011
Late Fall (LF)	Red clover incorporation	2 Nov. 2010	3 Nov. 2011
All	N_{min} 0–100 cm	May 19 2011	May 1 2012
Spring (SP)	Red clover incorporation	24 May 2011	2 May 2012
Early Fall + N_{70} (EF+N_{70})	Fertilizer application (70 kg N ha^{-1})	8 June 2011	24 May 2012
Hayed Late Fall + M_{70} (HLF+M_{70})	Liquid dairy manure application (70 kg N ha^{-1})	8 June 2011	24 May 2012
All	Wheat planting	8 June 2011	25 May 2012
All	Wheat plant N uptake measurement (boot stage)	10 Aug. 2011	18 July 2012
All	Wheat Harvest	20 Sep. 2011	4 Sep. 2012
All	N_{min} 0–100 cm	2 Nov. 2011	10 Nov. 2012

5.4.2 Spring wheat yield and N uptake

Grain yield from wheat plots was measured in Sep. of each year by plot combining a single 16 m x 1.2 m strip from the middle of each plot. Wheat uptake of N in the above ground parts of the plant was determined before head emergence (boot stage) on 10 Aug. 2011 (Year 1) and 18 July 2012 (Year 2). Aboveground biomass from five 1-m row lengths was harvested randomly from each plot (n=12) and dried (60°C; 72 to 96 hrs). Subsamples were then ground and composited before analyses for total C and N by dry combustion using a CN analyzer (Vario Max, Elementar, Germany).

5.4.3 N_2O and CO_2 soil emissions

Emissions of N_2O and CO_2 were measured in wheat plots from 1 June to 24 Aug. 2011 (Year 1) and 1 June to 17 Aug. 2012 (Year 2) with seven and six measurements in total, respectively. Measurements of CO_2 emissions are used in this context as a proxy for microbial activity, rather than looking at C as a GHG in a soil C budget.

A description of the chambers, method of gas sample collection, and analyses used in this study is given in Chapter 3. Cumulative CO_2 and N_2O emissions were calculated from June to Sep. in Year 1 and 2 (92-d) on an individual chamber basis by linear interpolation between sampling dates, using the assumption that gas flux measured on a sample date represents the daily average flux. Soil temperature (°C) and vol. water content ($m^3\ m^{-3}$) were measured (0–10 cm) in red clover (Mar.–June) and wheat plots (June and July) during gas flux measurements by recording the average of five individual readings adjacent to the soil flux chamber using standard hand held equipment. As a reminder, consideration needs to be given to the fact that all plants were removed from the inner collar area.

5.4.4 Soil properties

Profile (0–100 cm) soil inorganic N (N_{min}) was determined each year prior to amendment application in the spring (May) and post-harvest in the fall (Nov.) with the aid of a direct-push type, hydraulically powered soil sampler. At each sampling date, two cores were taken per sub-plot (i.e., rotation phase) (n=48) for a total of 96 cores. Cores were weighed for bulk density calculations and cut into 0 to 15, 15 to 30, 30 to 60, and 60 to 100 cm sections. Surface (0–15

cm) soil was sampled by compositing 10 soil samples per plot taken with a handheld probe (2 cm i.d.). Inorganic N was extracted with 0.5 M K_2SO_4 (3:1, liquid:soil) and filtered through glass fiber filter paper following the method of Keeney and Nelson (1982). Inorganic N (NH_4^+ and NO_3^-) was analyzed colorimetrically on an auto-analyzer (Technicon Instrument Corporation, 1971).

5.4.5 NO_3^- leaching

Drainage water was collected from each main plot (*n*=12) on a year-round basis in both Year 1 and 2. Drainage water was collected in 1-L water sampling bottles using an automated sampler (ISCO 6700 Portable Sampler, ISCO Inc., Lincoln, NE) designated for each block (*n*=3). Samplers were programmed to take a 500 mL water sample from each main plot daily or every 48 h during the fall of Year 1 and 2, and less frequently (minimum weekly) as drainage slowed due to soil frost, or water deficits. Sub-samples from the 500 mL water sample were collected into scintillation vials and immediately transferred to a freezer located in the same heated building as the water samplers. During seasons of low water drainage, water samples were taken manually from the spigot on each tile drain following precipitation or thawing events. Drainage flow from each main plot area (1200 m^2) was measured with tipping buckets beginning in the summer of 2011 and was used to calculate annual NO_3^- losses.

Before analysis, scintillation vials were thawed over-night at room temperature before colorimetric determination of NO_3^- as described above. The concentration of NO_3^- (mg NO_3^--N L^{-1}) was determined on a monthly basis by taking the mean from all sample dates during that month. Cumulative NO_3^- losses (NO_3^--N load) were calculated by summing the product of daily tile drainage volumes (m^3 ha^{-1}) and concentrations that were estimated between sample dates using linear interpolation.

5.4.6 Statistical analysis

Wheat N uptake, N concentration, grain yield, and cumulative emissions of CO_2 and N_2O were tested for differences in FMS (α=0.05) for each year using a mixed model with Block (*n*=3) as the random factor for all plant measurements and Sub-sample (*n*=6) for soil emissions. A similar analysis was used for monthly concentrations of NO_3^- (mg NO_3^--N L^{-1}) in drainage water and cumulative NO_3^- (kg N ha^{-1}) losses in each year using Block as the random factor.

Emissions of N_2O and CO_2 were analyzed for the effects of FMS and the interaction with Time across all sample dates using a mixed model repeated measures analysis that treated Sub-sample as the random factor in both Year 1 and 2. A similar repeated measure analysis was used for soil N_{min} ($NH_4^+ + NO_3^-$) data from the soil surface (0–15 cm), except Block was the random factor. Residual soil mineral nitrogen (RSMN) was assessed for effects on FMS, crop phase, and their interaction using a mixed model two-way analysis on soil sampled from the profile (0–100 cm) in Nov. and treating Block as the random factor. The effects of FMS, Year, and their interaction on monthly NO_3^- concentrations in drainage water were tested using a mixed repeated measures model that treated Block and Year as random factors. Soil profile N_{min} that was sampled in May from the wheat phase was evaluated for differences in FMS, Year, and their interaction using a mixed model that treated Block as the random factor. Data for CO_2 and N_2O emissions, RSMN, and N_{min} from the soil surface were $\log_{(10)}$ transformed prior to analysis to satisfy the assumptions of equal variance and normality. Following a significant FMS effect, means were separated using Tukey's mean separation test and a test of the simple effects (slices) when a significant interaction was detected. All effects were considered significant when $p \leq 0.05$, and statistical tests were performed using JMP software (SAS Institute Inc. Cary, NC, USA).

5.5 RESULTS

5.5.1 Precipitation, air temperature, and soil properties

Total precipitation (rain + snow) and monthly average air temperatures from the experimental site during the two years of study are presented in Table 5.2. Overall, cumulative precipitation was slightly higher (105 mm) in Year 1 with the majority of that difference falling in the spring. Average air temperatures were not different between years, with the exception of somewhat cooler temperatures beginning in Nov. through Jan. in Year 2. Monthly average temperatures were notably similar (± 0.5°C) from May to Oct. across years, with the exception of a cooler Aug. in Year 1. However, Year 1 was characterized by a much wetter early growing season (May to July), which also delayed wheat planting (Table 5.1).

Table 5.2. Sum of monthly precipitation and monthly mean air temperatures as measured at the experimental site over Year 1 (2011–12) and Year 2 (2012–13).

	Precipitation (mm)			**Mean air temperature (°C)**		
Month	Year 1	Year 2	30-yr (avg.)†	Year 1	Year 2	30-yr (avg.)
April	62	66	79	5.4	6.0	4.3
May	91	54	106	11.6	11.6	10.2
June	95	62	96	14.0	14.5	15.1
July	119	55	91	18.7	18.9	18.6
Aug.	159	129	90	18.7	20.1	18.2
Sep.	51	331	109	15.9	15.6	13.7
Oct.	326	53	108	10.3	10.4	8.0
Nov.	138	66	102	5.5	3.6	3.0
Dec.	128	101	74	1.2	–0.4	–3.6
Jan.	37	17	54	–4.4	–6.4	–6.7
Feb.	71	45	44	–4.9	–4.2	–6.1
Mar.	15	108	59	0.4	0.2	–1.8
Total	1291	1086	1011	7.7	7.5	6.1

† Climate normals (1981–2010) are from an Environment Canada weather station 20 km from the experimental site in Debert, NS.

Soil temperature (°C) and water content (m^3 m^{-3}) at the soil surface (0–10 cm) were measured from Mar. to July to estimate the potential for decomposition and N mineralization/nitrification processes (Fig. 5.1). Soil temperature increased earlier in Year 1, when compared to Year 2, which continued until June 1st when soil temperatures followed a similar pattern in both years. Soil water in Year 2 was high (>0.25 m^3 m^{-3}) at the beginning of spring (Apr.), but later declined to around 0.15 m^3 m^{-3} for the remainder of the sampling period. This was in contrast to soil water in Year 1, which began low and then remained at or above 0.25 m^3 m^{-3} for the majority of the sampling period.

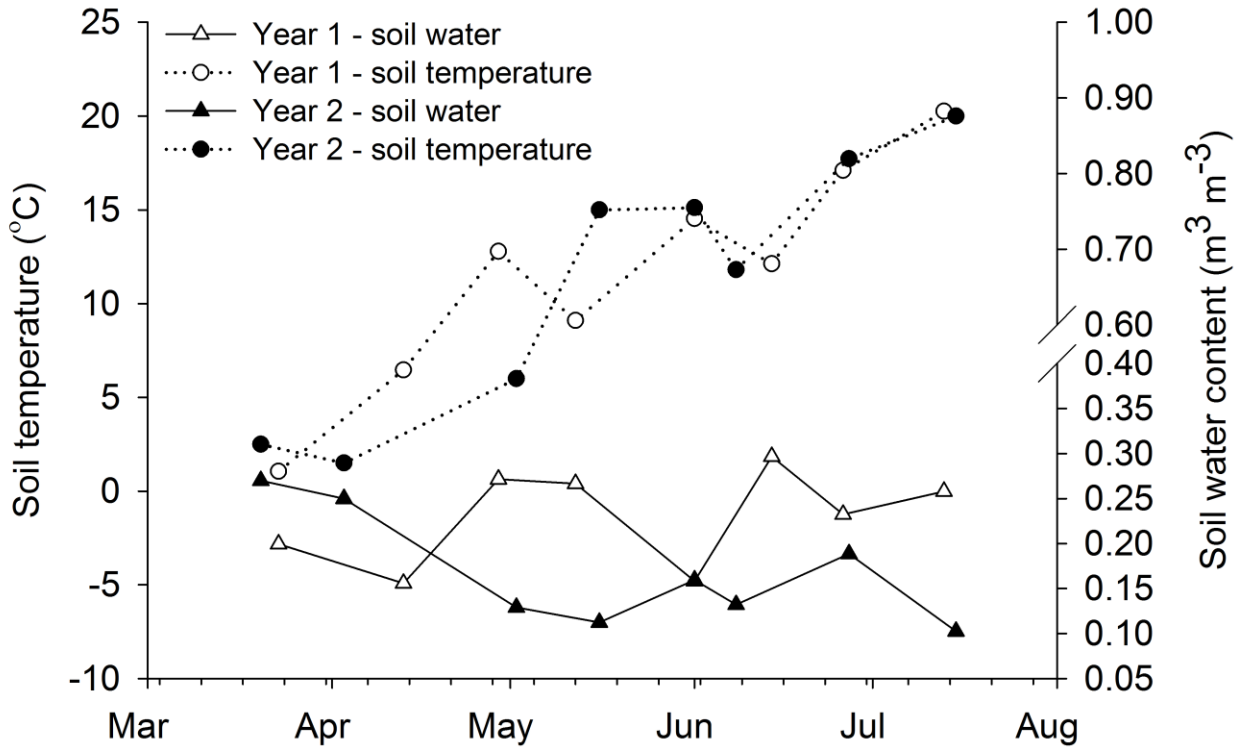

Fig. 5.1. Mean (*n*=12) soil water (m^3 m^{-3}) and temperature (°C) from the 0–10 cm depth in plots that are transitioning from red clover to spring wheat from Mar. 15 to July 15 over two years.

5.5.2 Wheat Yields and N Uptake

There were no differences in wheat grain yield (kg ha^{-1}) among FMS in Year 1 (p= 0.68) and Year 2 (p= 0.19), but annual average grain yields across all FMS were 1.6 times greater (p< 0.0001) in Year 2 (2683 kg ha^{-1}) as compared to Year 1 (1045 kg ha^{-1}). Whole plant N contents

(g N kg^{-1}) before head emergence were significantly different (p= 0.03) among FMS in Year 1 only (Table 5.3). The lowest tissue N concentration (g N kg^{-1}) was obtained for the HLF+M_{70} treatment, and the greatest for EF+N_{70}, with both LF and SP having intermediate values of 26.2 (g N kg^{-1}). There were no FMS treatment differences in whole plant N uptake (kg N ha^{-1}) at head emergence in Year 1 (p= 0.44) or Year 2 (p= 0.54). However, when mean N uptake across all treatments was compared between years (p< 0.0001), wheat plants had taken up 76% more N in Year 2 as compared to Year 1 and ranged from 1.2 to 1.9 kg N d^{-1} and 2.5 to 2.8 kg N d^{-1} in Year 1 and 2, respectively.

Table 5.3. Content and uptake of N by spring wheat prior to head emergence and spring wheat grain yield for Year 1 and 2 as affected by Fertility Management System (FMS). Acronyms for FMS are as follows EF+N_{70} (Early Fall + N_{70}), HLF+M_{70} (Hayed Late Fall + M_{70}), LF (Late Fall), and SP (Spring).

FMS	Tissue N (g N kg^{-1})		Plant N uptake (kg N ha^{-1})		Wheat grain yield (kg ha^{-1})	
	Year 1	Year 2	Year 1	Year 2	Year 1	Year 2
EF+N_{70}	30.8 (0.9)† a‡	25.0 (3.0)	121.63 (20.2)	176.43 (14.2)	867 (95.7)	2246 (393)
HLF+M_{70}	25.4 (2.5) b	24.8 (2.1)	92.53 (25.8)	158.54 (14.5)	1073 (243)	2843 (222)
LF	26.2 (0.8) ab	23.9 (1.6)	90.22 (14.8)	167.11 (15.8)	1129 (52.2)	2794 (79.0)
SP	26.2 (1.2) ab	27.5 (1.5)	73.68 (11.6)	163.7 (3.6)	1110 (120)	2626 (307)
p-value	0.03	0.91	0.44	0.54	0.68	0.19

† SEM (standard error of the mean) n=6

‡ FMS with the same letter in a column are not different from one another (p > 0.05)

5.5.3 N_2O and CO_2 emissions

Cumulative N_2O emissions (kg N_2O ha^{-1}) in Year 1 (Table 5.4) were greatest (p= 0.04) from HLF+M_{70} plots (0.872 kg N_2O-N ha^{-1}) as compared to all other FMS that had a range of 0.057 to 0.164 kg N_2O-N ha^{-1}. In Year 2, cumulative N_2O emissions were not different (p= 0.44) among the four FMS, which had a range of 0.216 to 0.574 kg N_2O-N ha^{-1}. There was no effect of FMS in Year 1 (p= 0.19) when N_2O emissions were considered across all sample dates with a range of emissions of 0.57 to 9.34 g N ha^{-1} d^{-1} (Fig. 5.2 A). However, the FMS x Time interaction was significant (p< 0.001). The greatest emission events occurred on 14 Jun. 2011 in

the HLF+M_{70} treatment with 38.95 g N_2O-N ha^{-1} d^{-1} and another similar event a month later that was less than 20 g N_2O-N ha^{-1} d^{-1}. In Year 2, there were neither effects of FMS (p= 0.22) on N_2O emissions (Fig. 5.2 B) nor interaction with Time (p= 0.5), but mean emission values across all sample dates had a range of 1.99 to 6.74 g N_2O ha^{-1} d^{-1}.

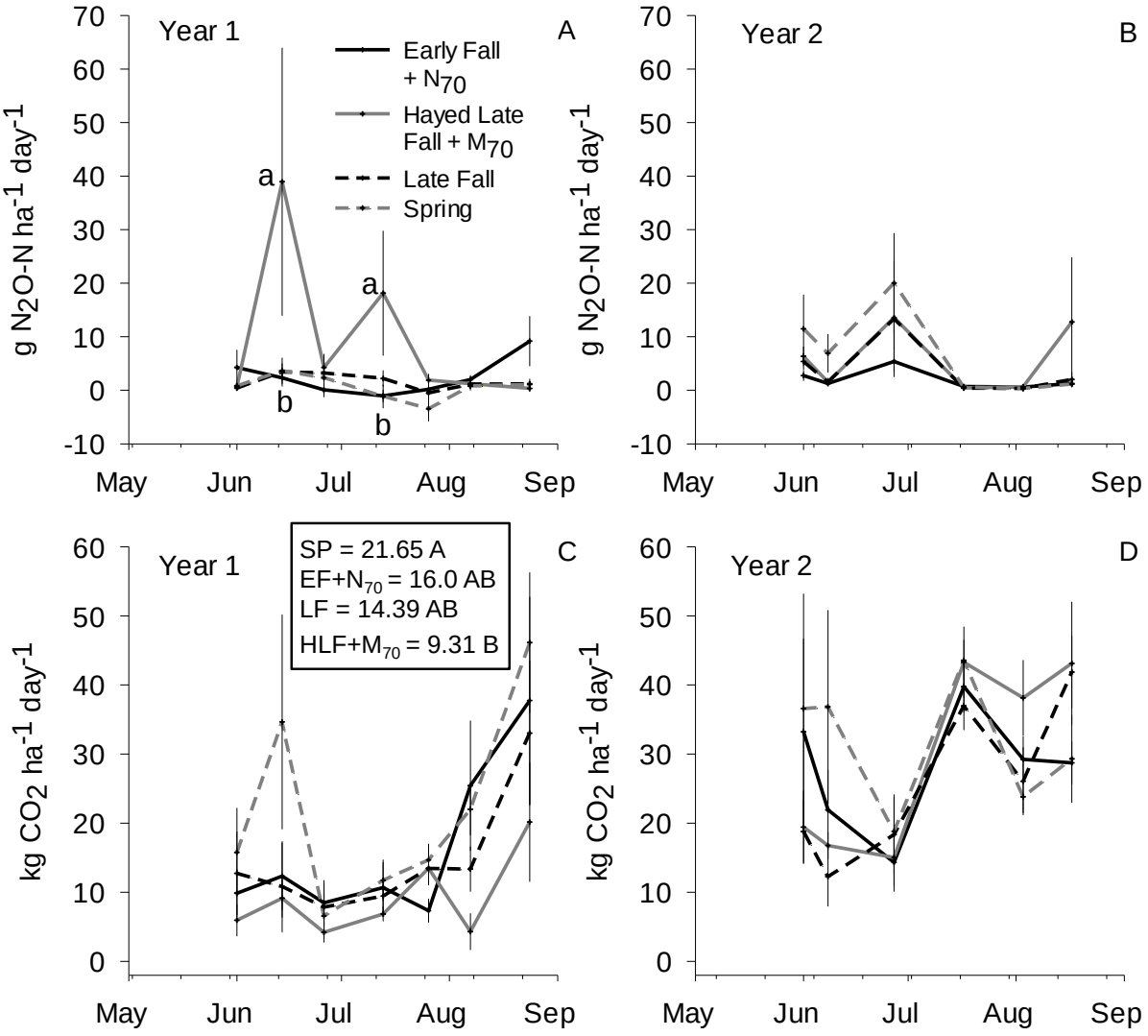

Fig. 5.2. N_2O (A, B) and CO_2 (C, D) emissions from the wheat phase (June–Sep.) of the crop rotation over 2 years. Dates with a significant interaction (FMS x time) in N_2O and CO_2 daily emissions are separated for differences in fertility management systems (FMS) with different lower case letters ($p \leq 0.05$). Significant effects of FMS across all sample dates are indicated in a text box with different upper case letters ($p \leq 0.05$) in panel C only as effects of FMS across dates were not significant in the other analyses. Acronyms for FMS are as follows: EF+N_{70} (Early Fall + N_{70}), HLF+M_{70} (Hayed Late Fall + M_{70}), LF (Late Fall), and SP (Spring). Error bars represent the SEM (n=6).

Across all sample dates in Year 1, CO_2 emissions (kg CO_2 ha^{-1} d^{-1}) were highest (p= 0.02) in SP plots (21.65 kg CO_2 ha^{-1} d^{-1}) as compared to HLF+M_{70} (9.31 kg CO_2 ha^{-1} d^{-1}). Intermediate values were measured from EF+N_{70} and LF of 16.00 and 14.39 kg CO_2 ha^{-1} d^{-1}, respectively (Fig. 5.2 C). There was no FMS x Time interaction (p= 0.54) in Year 1. In the second year of study (Fig. 5.2 D), there was no effect of FMS on CO_2 emissions (kg CO_2 ha^{-1} d^{-1}) across all sample dates (p= 0.83), nor was the FMS x Time interaction significant (p= 0.47). When overall CO_2 emissions from all treatments and sample dates are compared, there was an increase of 87% in Year 2 as compared to Year 1. Cumulative CO_2 emissions (kg CO_2 ha^{-1}) in Year 1 (Table 5.4) were lowest (p= 0.0001) from the HLF+M_{70} system (874.03 kg CO_2 ha^{-1}), followed by EF+N_{70} and LF that had a mean value of 1436.81 kg CO_2 ha^{-1} when compared to the SP system (2036.48 kg CO_2 ha^{-1}). There were no differences (p= 0.62) among FMS for cumulative CO_2 emissions in Year 2.

Table 5.4. Cumulative CO_2 and N_2O-N (kg ha^{-1}) emissions among fertility management systems (FMS) from the wheat phase during 1 June to 1 Sep in each year. Acronyms for FMS are as follows: EF+N_{70} (Early Fall + N_{70}), HLF+M_{70} (Hayed Late Fall + M_{70}), LF (Late Fall), and SP (Spring).

FMS	Year 1		Year 2	
	CO_2	N_2O-N	CO_2	N_2O-N
	June to Sep. cumulative emissions (kg ha^{-1})			
EF+N_{70}	1504 (369)† ab‡	0.094 (0.09) b	2564 (337)	0.216 (0.04)
HLF+M_{70}	874 (159) b	0.872 (0.49) a	2878 (291)	0.574 (0.33)
LF	1369 (236) ab	0.164 (0.03) b	2545 (256)	0.359 (0.24)
SP	2036 (147) a	0.057 (0.08) b	2788 (437)	0.573 (0.22)
p-value	**0.01**	**0.04**	0.86	0.51

† SEM (standard error of the mean) n=6

‡ FMS with the same letter in a column are not different from one another ($p > 0.05$)

5.5.4 Soil inorganic N

Surface (0–15 cm) N_{min} content (kg N ha^{-1}) measured in Year 2 clover (Apr.–June) and wheat (June–Aug.) rotation phases are presented for both study years (Fig. 5.3). When the mean for each FMS was taken across all sampling dates, an overall effect (p= 0.047) was seen in Year

1 for FMS. Mean N_{min} content was highest for EF+N_{70} (44.92 kg N ha^{-1}) when compared to the lowest in SP (21.01 kg N ha^{-1}), while the mean of LF and HLF+M_{70} was intermediate (27.69 kg N ha^{-1}). When differences in FMS were evaluated for each sampling date (p= 0.029), plots that received supplemental N (i.e., EF+N_{70}, HLF+M_{70}) were significantly greater than SP and LF through the month of June by 29 to 37 kg N ha^{-1}. In the month of July, EF+N_{70} was greater than all other FMS on 13 and 26 July 2011 with 81 and 38 kg N ha^{-1}, respectively. The unamended FMS (i.e., LF and SP) resulted in consistently low (<40 kg N ha^{-1}) N_{min} levels when compared to FMS that were supplemented with manure and inorganic fertilizer.

In the second year of study, there were no FMS effects (p= 0.68), but the FMS x sampling date interaction was significant (p< 0.0001). Both of the Nov. incorporated clover FMS (i.e, LF and HLF+N_{70}) had N_{min} contents that were double those in EF+N_{70} and SP on 23 Apr. 2012. The smallest N_{min} content on 24 May 2012 was from EF+N_{70} as compared to all other FMS. Following wheat planting (i.e., 25 May 2012), the unamended FMS (i.e., EF, LF) had surface N_{min} contents that were 36 and 59% smaller than the mean of amended FMS (i.e., EF+N_{70} and HLF+M_{70}) on 1 and 8 June 2012, respectively. This trend continued until the middle of July when soil N_{min} was < 20 kg ha^{-1} in all FMS.

Soil profile (0–100 cm) N_{min} was evaluated from red clover plots in May, prior to SP plowdown and wheat planting, which resulted in no overall effect of FMS (p= 0.49) (Fig. 5.4), nor was the FMS x Year interaction significant (p=0.53). However, there was a dramatic difference between years (p< 0.0001) where there was nearly four times the profile N_{min} in Year 1, as compared to Year 2 (24.77 kg N ha^{-1}). In Year 1, a large proportion of N_{min} within the profile was below 60 cm (57%), while only as 27% could be found within the top 0–15 cm; this is evidence of substantial mineralization and nitrification before the sampling date, as well as the downward movement of NO_3^- in the spring of Year 1.

There was a striking difference between N_{min} measured at the soil surface (Fig. 5.3) and N_{min} measured within the soil profile (Fig. 5.4) in Year 2. Approximately, 30 kg N ha^{-1} was measured in the upper soil horizon (0–15 cm) on 23 Apr. 2012 (Fig. 5.3), yet only ~26 kg N ha^{-1} across all FMS in the entire soil profile (0–100 cm) on 1 May 2012 (Fig. 5.4). Heavy precipitation (42 mm) during the week between sampling dates, and a difference in soil sampling methodology (i.e., composite sample vs. deep soil core) could be possible explanations for the discrepancy.

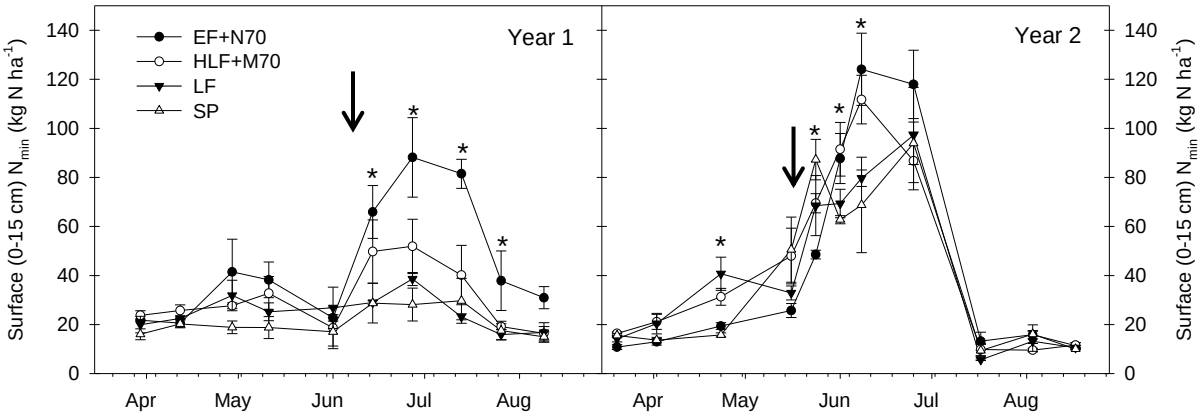

Fig. 5.3. Inorganic N (NH_4^+ + NO_3^-) from the 0–15 cm soil depth for Year 1 (A) and Year 2 (B). Dates with a significant interaction (FMS x time) are indicated with an asterisk (p< 0.05). Acronyms for FMS are as follows EF+N_{70} (Early Fall + N_{70}), HLF+M_{70} (Hayed Late Fall + M_{70}), LF (Late Fall), and SP (Spring). Arrows indicate wheat planting. Error bars represent the SEM (n=3).

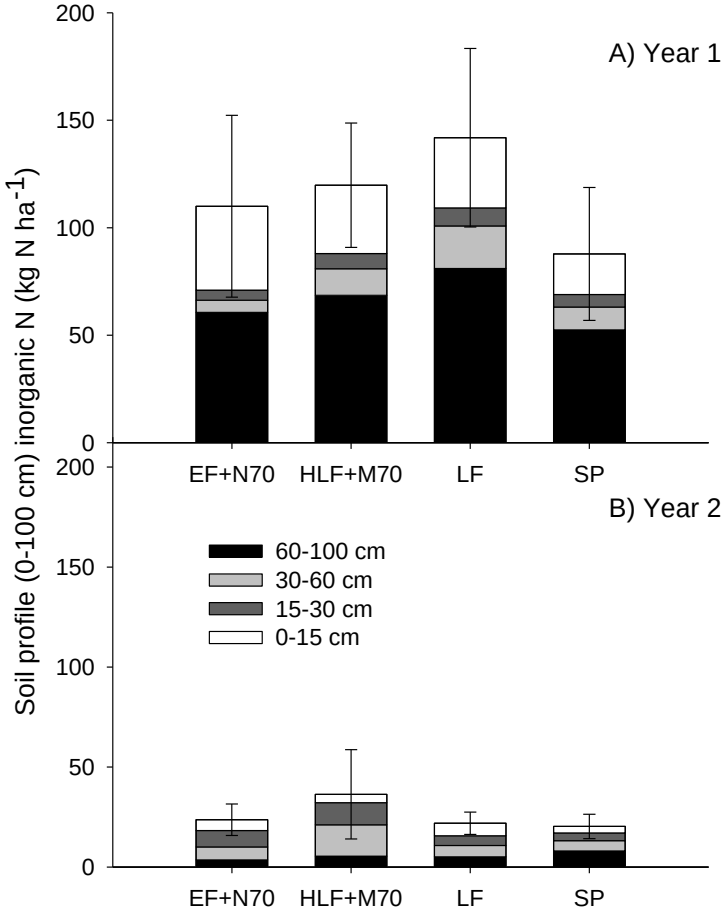

Fig. 5.4. Inorganic N (N_{min}) within the soil profile (0–100 cm) from red clover plots prior to wheat planting and sampled on (A) 19 May 2011 and (B) 1 May 2012. Categories along the x-axis refer to Early Fall + N_{70} (EF+N_{70}), Hayed Late fall + M_{70} (HLF+M_{70}), Late Fall (LF), and Spring (SP) Fertility Management Systems (FMS).

5.5.5 NO_3^- leaching and fall residual Soil N

Differences in monthly NO_3^- concentrations and total NO_3^- losses in drainage water among FMS were assessed using repeated measures analysis in each year of study (Fig. 5.5). There were no differences in FMS in Year 1 (p= 0.19) or Year 2 (p= 0.09), nor were there any significant interactions with Month in the first (p= 0.59), or second (p= 0.98) years of study. Similarly, cumulative annual NO_3^- losses were not significant across FMS in Year 1 (p= 0.16) or Year 2 (p= 0.38) with a range of 13.7 to 32.1 and 10.5 to 20.6 kg N ha^{-1}, respectively.

However, when drainage water NO_3^- concentration among FMS was evaluated in a two-way analysis (i.e., FMS, Year, and their interaction) for each month there was one consistent effect (Table 5.6); EF+N_{70} had the highest concentration of NO_3^- in drainage water in Oct., Nov., Dec., and May with values of 13.78, 10.49, 7.85, and 10.69 mg N L^{-1}, respectively. Because drainage water is collected from the entire 4-yr crop rotation, it is appropriate to evaluate monthly mean NO_3^- concentrations over the 2-yr study, rather than direct effects of FMS in each year. Seasonal trends are similar in each year, as indicated in Fig. 5.5, but overall the highest concentrations were consistently measured from EF+N_{70} during the months of Oct., Nov., Dec., and May. There were no significant interactions between FMS and Year during any month.

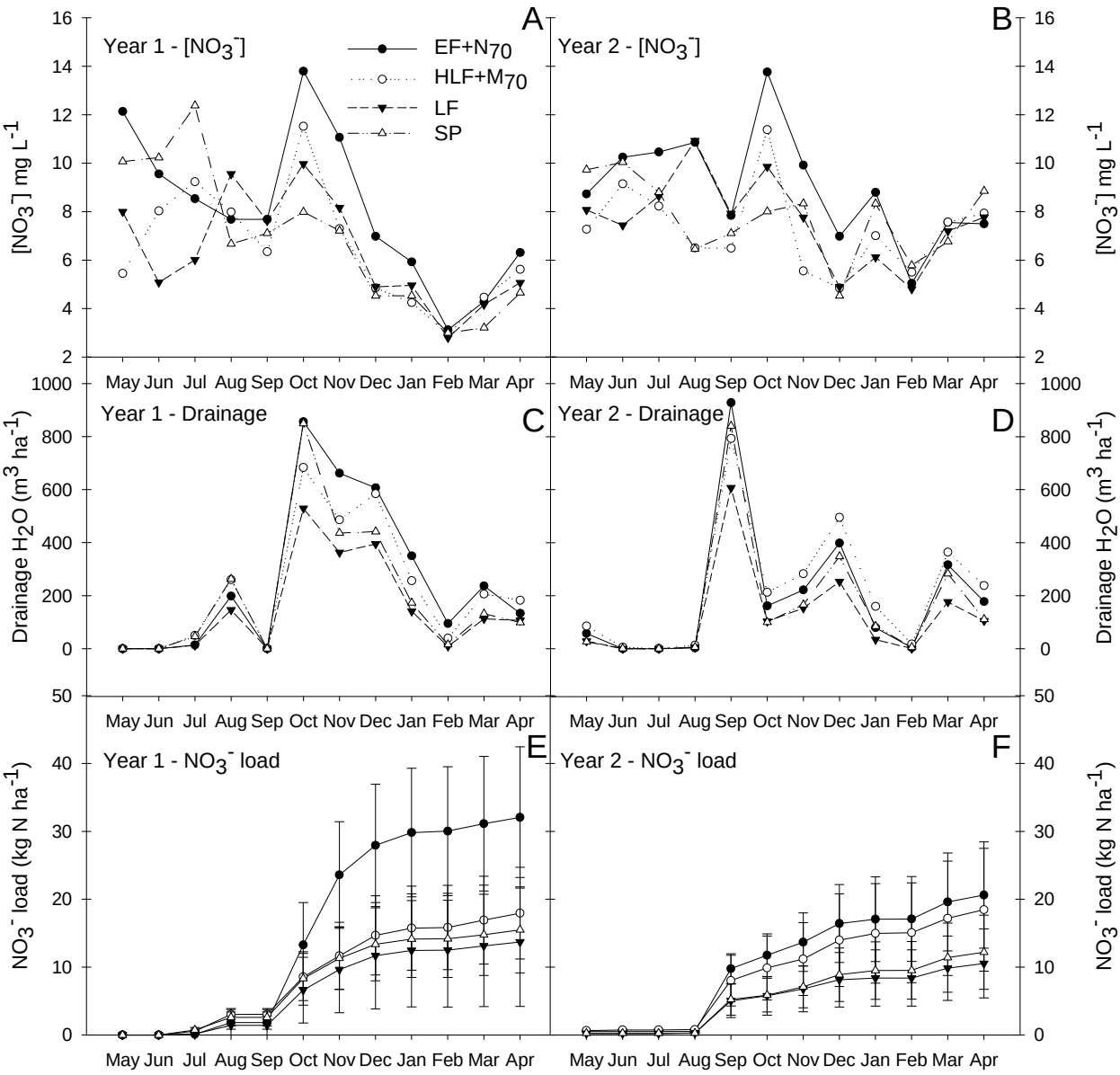

Fig. 5.5. Monthly NO_3^- concentration in drainage water during Year 1 (A) and 2 (B). Drainage water volumes (m^3 ha^{-1}) for each FMS in both years (C, D). Cumulative NO_3^- losses (NO_3^- Load) from May through Apr. (E, F). Acronyms for FMS are as follows EF+N_{70} (Early Fall + N_{70}), HLF+M_{70} (Hayed Late Fall + M_{70}), LF (Late Fall), and SP (Spring). Error bars represent the SEM (*n*=3).

Table 5.5. Concentration of NO_3^- in drainage water among Fertility Management Systems (FMS) averaged over two years Acronyms for FMS are as follows $EF+N_{70}$ (Early Fall + N_{70}), $HLF+M_{70}$ (Hayed Late Fall + M_{70}), LF (Late Fall), and SP (Spring).

	June	**July**	**Aug.**	**Sep.**	**Oct.**	**Nov.**	**Dec.**	**Jan.**	**Feb.**	**Mar.**	**April**	**May**
FMS	- mg NO_3^--N L^{-1} -											
$EF+N_{70}$	9.9	9.5	9.27	7.76	13.78 a[†]	10.49 a	7.85 a	7.36	4.08	5.93	6.91	10.69 a
$HLF+M_{70}$	8.59	8.72	7.25	6.42	11.45 b	6.42 b	6.45 ab	5.63	4.22	6.02	6.78	6.08 b
LF	6.26	7.34	10.24	7.77	9.91 b	7.96 b	6.1 ab	5.54	3.8	5.68	6.42	7.63 ab
SP	10.13	10.57	6.57	7.11	8.0 c	7.78 b	5.28 b	6.43	4.39	4.99	6.86	9.36 ab
Source of variation	- Probability (*p*) -											
FMS	0.49	0.38	0.56	0.82	**< 0.001**	**< 0.001**	**0.009**	0.07	0.81	0.47	0.96	**0.001**
FMS x Year	0.97	0.43	0.77	0.99	0.99	0.08	0.49	0.31	0.94	0.75	0.61	0.21
n =	6	6	6	6	6	6	6	6	6	6	6	6
SE	1.64	2.26	2.16	1.4	1.85	1.43	1.15	1.43	1.25	1.69	1.82	0.85

† Values within a column with the same letter are not significantly different at the 0.05 level of probability.

Fall RSMN was determined in Nov. sampled soil cores (0–100 cm) in both years (Table 5.6). Both the FMS (p=0.001) and the rotation phase (p=0.045) effects were significant in Year 1, but their interaction was not (p=0.12). Highest N_{min} contents from the entire profile (0–100 cm) were from EF+N_{70} and LF with 33.44 and 29.49 kg N ha^{-1}, respectively, while HLF+M_{70} was the smallest (20.29 kg N ha^{-1}) and SP was intermediate with 26.14 kg N ha^{-1}. The greatest N_{min} content among rotation phase in Year 1 was measured from spring wheat (Y4; 29.53 kg N ha^{-1}) as compared to the first red clover (Y2) phase (22.04 kg N ha^{-1}), while soybean (Y1) and the second red clover (Y3) phase of the rotation were intermediate with a mean value of 28.9 kg N ha^{-1}.

In the second year of study, the effect of FMS was marginally significant (p=0.054), but profile N_{min} was again highest from EF+N_{70} (53.47 kg N ha^{-1}) as compared to all other FMS (39.54 kg N ha^{-1}). The effect of rotation phase was not significant (p=0.147), but the interaction with FMS was (p=0.006). Differences among FMS in the second red clover phase (Y3) of the rotation were significant and followed the same pattern as previously discussed for the main effects. The significant interaction was due to differences in crop phase within HLF+M_{70}; measuring 80% more N_{min} in the spring wheat and the first red clover phase (Y2), as compared to soybean and the second red clover year (27.86 kg N ha^{-1}).

Table 5.6. Fall residual soil mineral N (RSMN) within the soil profile (0–100) cm. Soil sampled on 2 Nov. 2011 (Year 1) and 10 Nov. 2012 (Year 2). Acronyms for FMS are as follows: EF+N_{70} (Early Fall + N_{70}), HLF+M_{70} (Hayed Late Fall + M_{70}), LF (Late Fall), and SP (Spring).

	Year 1					Year 2				
	Soybean (Y1)	Red clover (Y2)	Red clover (Y3)	Spring wheat (Y4)	**Across rotation phase**	Soybean (Y1)	Red clover (Y2)	Red clover (Y3)	Spring wheat (Y4)	**Across rotation phase**
FMS	**Crop rotation phase**					**Crop rotation phase**				
EF+N_{70}	31.93 (8.6)†	22.8 (4.8)	48.12 (9.62)	30.89 (7.55)	**33.44 (9.11) A‡**	37.12 (10.03)	61.18 (21.86)	62.38 (14.39) A	54.21 (11.29)	**53.47 (15.45) A**
HLF+M_{70}	19.6 (3.34)	19.3 (1.96)	17.1 (2.43)	25.17 (3.15)	**20.29 (3.13) B**	32.13 b (5.17)	45.96 a (8.88)	23.58 b (5.72) B	54.22 a (13.15)	**38.97 (10.81) B**
LF	36.7 (15.58)	22.08 (4.75)	27.6 (6.76)	31.6 (3.38)	**29.49 (8.95) A**	52.41 (18.21)	35.45 (7.35)	25.82 (5.7) B	52.37 (8.69)	**41.51 (12.35) B**
SP	26.02 (1.43)	23.97 (5.76)	24.12 (4.93)	30.47 (9.04)	**26.14 (5.75) AB**	45.41 (6.18)	33.38 (7.25)	42.08 (10.79) B	31.66 (8.42)	**38.13 (8.49) B**
Across FMS	**28.56 (9.28) a§**	**22.04 (4.36) b**	**29.24 (9.11) a**	**29.53 (6.09) a**		**41.77 (11.36)**	**43.99 (13.66)**	**38.47 (12.99)**	**48.12 (11.33)**	

Source of variation	Probability (*p*)			
FMS	Year 1	**0.001**	Year 2	**0.054**
Rotation phase (RP)		**0.045**		0.147
FMS x RP		0.116		**0.006**

† Values in brackets represent the SEM (*n*=3).

‡ Differences in Fertility Management Systems (FMS) are indicated in each year with different upper case letters that are significantly different at the 0.05 level of probability. Bolded upper case letters indicate main effects (across rotation phase) and plain text letters are for the FMS x RP interaction.

§ Differences in Rotation Phase (RP) are indicated in each year with different lower case letters that are significantly different at the 0.05 level of probability. Bolded lower case letters indicate main effects (across FMS) and plain text letters are for the FMS x RP interaction.

5.6 DISCUSSION

5.6.1 Soil N supply from GMr

Although there were no differences in wheat grain yield or N uptake among FMS in either year of study, the grain yield increase in Year 2 as compared to Year 1 across all management systems suggests that the SNS was less synchronized with plant uptake in Year 1. Across all FMS and sample dates, N_{min} (kg N ha^{-1}) was 42% greater in Year 2 as compared to Year 1. The majority of this difference was seen during the month of June when N_{min} was 39 and 91 kg N ha^{-1} in Year 1 and 2, respectively. The reduction of surface soil N_{min} to low levels (~20 kg N ha^{-1}) also occurred earlier in the season in Year 2 as compared to Year 1, an indication of greater plant uptake in Year 2, as discussed previously. However, grain yields in Year 2 were comparable to other organic systems (Kaut et al., 2009), as well as the 50-yr average (2.6 Mg ha^{-1}) from a fertilized fallow–wheat system in Saskatchewan, Canada (Campbell et al., 2011). Loes et al. (2011) described a 10-fold difference in barley grain yield over a 4-yr study and attributed the low yields to cold and wet soil conditions early in the growing season at two sites with similar soil texture and bulk density in Norway. Lynch et al. (2008) in previous work at the same field site as our study attributed a tuber yield reduction to a soil water deficit at the Brookside experimental site, which is an example of the influence that weather can have on the temporal variation of soil temperature and water content, and ultimately the SNS.

The bulk of the in-season SNS is derived from newly mineralized N, rather than from a carry-over effect from the fall (Zebarth et al. 2009), which makes agricultural systems that are reliant on organic N inputs from GMr vulnerable to reductions in the SNS. Wheat N uptake was at least 70% greater in Year 2 as compared to Year 1 from FMS that were reliant on organic N sources (i.e., LF, HLF+M_{70}, SP) (Table 5.3). In contrast, N uptake in Year 2 was only 45% greater when supplemental N was applied as inorganic fertilizer and clover was incorporated in Sep. (EF+N_{70}). The difference between FMS using organic N sources and EF+N_{70} is further evidence that the timing for the majority of the SNS from the legume GMr in Year 1 was before

the peak N uptake time of the wheat crop. Furthermore, emissions of CO_2, a measure of C availability, on 14 Jun. 2011 were highest from the SP incorporated clover system three weeks following incorporation, and then decreased to similar values measured in the other three systems (5 to 10 kg CO_2 ha^{-1} d^{-1}) 12 days later. Reduced CO_2 emissions from all FMS in Year 1, as compared to Year 2, is again more evidence of low heterotrophic respiration that could be explained by the observed cool and wet soil conditions in early spring of Year 1. Although there has been extensive work in recent years to improve the prediction of SNS there is still a need for regional models, as no single approach has been successful across all soils (St. Luce et al., 2011). Dessureault-Rompre et al. (2015) developed a kinetic model using a variety of soils from potato cropped fields in Atlantic Canada that were used to compare the predicted in-season SNS with actual plant uptake and RSMN with the goal of using simple soil and weather measurements to improve fertilizer recommendations in conventionally managed operations. Although the model was not developed with GMr systems in mind, there is considerable advancements being made with respect to SNS in humid climates that will likely further our understanding of the rate and timing of SNS in GMr systems. Yield reductions in GMr systems could be avoided if early season SNS dynamics were better understood by adjusting the use of supplemental N based on specific soil types, history, and weather patterns.

5.6.2 NO_3^- leaching

Precipitation was 55% greater from Apr. through July in Year 1, as compared to Year 2 (Table 5.2), which indicates that there was more soil water in Year 1 and a larger potential for spring NO_3^- leaching from the increased drainage. Unfortunately, we were not measuring drainage flow from Apr. to July, but NO_3^- contents above 50 kg N ha^{-1} below the 60 cm soil depth on 19 May 2011 (Fig. 5.4) imply that there was considerable leaching at this time. Similarly, 46 kg N ha^{-1} were leached from red clover–spring barley rotation on a sandy loam in Sweden (Bergstrom and Kirchmann, 2004), or 71 kg N ha^{-1} on a coarse sand in Denmark that utilized a variety of GMr rotations (Askegaard et al., 2011). The majority of NO_3^- lost to leaching occured during the fall, as opposed to during the growing season, which is in agreement with previous studies (Belanger et al., 2003; Lynch, 2009; Jiang et al., 2011). If the SNS is unsynchronized with crop uptake in GMr systems, there will be a large residual soil NO_3^- pool that is susceptible to N losses. The risk for NO_3^- leaching is even greater when the FMS uses

mineral fertilizer as a supplemental N source, as was measured from EF+N_{70} when monthly NO_3^- concentrations in drainage water were considered (Table 5.5). However, it is not clear if this was a result of the earliest fall clover incorporation date or the use of supplemental fertilizer in the successive wheat crop. The EF+N_{70} FMS had the highest RSMN levels across all phases of the rotation in both years of study, in addition to having the highest RSMN from the red clover (Y3) phase of the rotation in Year 2 (Table 5.6). Considering the additional energy required to produce and transport inorganic fertilizers in addition to the apparent increase in NO_3^- leaching losses, there is another negative aspect of the EF+N_{70} FMS when compared to other GMr incorporation dates and supplemental N use examined in this study.

Lahti and Kuikman (2003) conducted a similar experiment that investigated multiple timings of vetch incorporation (Sep., Oct., Dec., and May) in Finland for a successive spring wheat crop. They found that soil N_{min} increased earlier according to how early a leguminous GMr is incorporated in the fall, which poses a greater risk for NO_3^- leaching before the cash crop phase. Askegaard et al. (2011) reported that greater NO_3^- leaching occurred when the frequency of fall field tillage was increased in addition to earlier field tillage in the fall. We had a similar result where higher NO_3^- concentrations were measured in tile drainage water from the EF+N_{70} FMS in Oct., Nov., Dec., and May. A soil N supply that was unsynchronized with plant uptake, as discussed above, is also likely the cause of the increased total NO_3^- losses in drainage water in Year 1 as compared to Year 2 (Fig. 5.5). The removal of clover biomass as hay, and the subsequent application of liquid dairy manure (HLF+M_{70}), had no observable effect on NO_3^- leaching when compared to the other treatments. This result was supported by no difference in surface N_{min} and RSMN within the soil profile (0–100 cm), which was in agreement with a study conducted by Askegaard et al. (2005). Considering this, GMr incorporation in Sep. (e.g., EF+N_{70}) is not recommended unless an adequate cover crop can be established immediately after planting, as described by Thorup-Kristensen et al. (2003) and Rasouli et al. (2014).

5.6.3 N_2O emissions

Cumulative N_2O emissions (92 d) from our study are overall quite low, with means of 0.3 and 0.43 kg N_2O-N ha^{-1} across treatments for Year 1 and 2, respectively. Emissions were comparable to an unfertilized continuous corn experiment, described by Drury et al. (2014a), with N_2O emissions of 0.51 kg N ha^{-1} from Woodslee, ON. However, emissions were much

lower than a fertilized continuous corn treatment (7.4 kg N ha^{-1}) also described by Drury et al. (2014a), or from a corn crop in rotation with alfalfa (5.8 kg N_2O-N ha^{-1}) in Rock Springs, PA (Adviento-Borbe et al., 2010). Nitrous oxide emission events were small and short lived among all FMS in Year 1 and 2, with the exception of higher emissions being measured from the HLF+M_{70} system during two sampling dates in Year 1 following manure application. The highest daily emission occurred on 14 June 2011 in the HLF+M_{70} FMS, six days following manure application, yet there was no simultaneous peak in CO_2 emissions, implying that anoxic conditions were present due to a water content of 0.5 m^3 m^{-3} at the time of gas measurement at those specific field locations (data not shown). Water content on 14 June 2011 was on average 0.29 m^3 m^{-3} across all 24 gas-sampling locations, as there were several days of rain prior to and during this sampling event. The localized "hot spot" phenomenon for N_2O emissions was evident during this sampling event, as the influence of manure application together with the occurrence of above average water contents in these plots created conditions for increased N_2O emissions. The fact that there were no N_2O emission events in Year 2 following manure application on 24 May 2012 from LF+M_{70} (Fig. 5.2) is another indication of soil conditions in Year 1 that were less than ideal for growth in the wheat crop, and the subsequent uptake of N_{min}. The majority of N_2O emissions following manure application occur within 40 d (Rochette et al., 2008; Chantigny et al., 2010) and are typically associated with high WFPS following precipitation events, in addition to the additional water contained within the slurry (Mkhabela et al., 2008). Rochette et al. (2004) measured a large N_2O emission (830.4 g N_2O ha^{-1} d^{-1}) three weeks following pig slurry application (200 kg total-N ha^{-1}) in Quebec, Canada. This emission event was attributed to denitrification due to an increase in WFPS, and the abundance of NO_3^- and available C.

Red clover, grown in Atlantic Canada in a 2-yr rotation with fertilized (193 kg N ha^{-1}) potato, produced more cumulative (May to Nov.) N_2O emissions (1.41 kg N ha^{-1}) than any other preceding crop (e.g., barley, canola, soybean); the majority of these emissions occurred immediately after potato fertilization and planting (Snowdon et al., 2013). In our study, we did not see any response in N_2O emissions with fertilizer application in either year of study; however, our fertilizer application rate of 70 kg N ha^{-1} was much less than that used by Snowdon et al. (2013). Furthermore, yield-scaled N_2O emissions were determined in a meta-analysis to be minimal when N surplus (i.e., applied N minus crop uptake) was in the range of 0 to 50 kg N ha^{-1} (Van Groenigen et al., 2010). Considering we had no yield or wheat N uptake response to FMS

in either year (Table 5.3), we can assume that a moderate fertilizer application of 70 kg N ha^{-1} in addition to the incorporation of red clover the previous Sep. was not in excess of crop demand. Therefore, it is not surprising that we did not see a response in N_2O emissions from this system.

5.6.4 Summary of Soil NO_3^-, N uptake, and N losses to air and drainage water

In temperate humid environments, such as those found in Atlantic Canada, the SNS through the growing season is from in-season N mineralization, rather than N_{min} carried over from the previous growing season. Year to year weather variations created soil conditions in early spring of Year 1 that unexpectedly mineralized N_{min} at a high rate and was followed by intense precipitation that distributed N_{min} deep through the soil profile (Table 5.7). Wet soil conditions delayed wheat planting until 8 June 2011, and much of the SNS occurred too early in the growing season; as indicated by the high spring profile N_{min} pool, and the low wheat N uptake on 10 Aug. 2011. Wheat N uptake was by far the largest N_{min} pool in the second study year, and was 76% higher than Year 1.

Although clover removal and the application of dairy manure (i.e., HLF+M_{70}) produced the largest emission of N_2O when compared to all FMS in any year, in-season N_2O emissions were low in comparison to other studies (e.g., Rochette et al., 2008; Chantigny et al., 2010; Ball et al., 2014). Furthermore, the smallest in-season N_2O emissions in each year were measured from the FMS that had received a moderate application of mineral fertilizer (i.e., EF+N_{70}), yet had the highest annual leaching losses when compared to the other FMS, which were unexpected results.

Nitrogen losses from NO_3^- leaching are most common through the fall, but fall soil profile N_{min} pool (RSMN; Table 5.6) was within 25.17 to 54.22 kg NO_3^--N ha^{-1}, which indicated a low potential for annual NO_3^- leaching in these GMr systems. Fall RSMN in this study was within the average range for arable soils in Quebec and Ontario from 1981 to 2006 (De Jong et al., 2009). Again, N losses as NO_3^- in drainage tiles were low overall from all FMS, but we were unable to measure NO_3^- leaching losses during the spring of 2011. With the majority of the N_{min} below the rooting depth on 19 May 2011, and an additional 100 mm of precipitation over the next 42 days, there were likely large N losses as NO_3^- in drainage water that had gone unaccounted. In environments that lack extend periods of frozen soil conditions fall plowing is typically avoided, in favor of spring or late winter GMr incorporation (Ball et al., 2007).

However, in our study there was evidence of extensive N leaching losses in spring of Year 1 and slow wheat growth, as indicated by lower CO2 emissions, which likely contributed to the lowered overall wheat yields in all FMS. Surprisingly, supplemental N additions prior to wheat planting did not help alleviate the plant-available soil N deficit that occurred early in the growing season, another indication of soil conditions that were not favorable for rapid wheat growth early in the growing season of Year 1.

Table 5.7. Summary table of the different N_{min} pools from fertility management systems (FMS) over two years. Spring soil profile N_{min} was measured following fall incorporated GMr plots, but before SP incorporation. Fall soil profile N_{min} is measured from wheat plots following harvest. Mean N_{min} across FMS for each column is referred to as total with the SD in brackets.

FMS	Spring soil profile N_{min}	Wheat N uptake	In season N_2O emissions	Fall soil profile N_{min}	Annual NO_3^- leaching
	- kg N ha^{-1} -				
	- Year 1 -				
EF+N_{70}	109.97	121.63	0.094	30.89	32.05
HLF+M_{70}	119.76	92.53	0.872	25.17	17.94
LF	141.88	90.22	0.164	31.6	13.69
SP	87.81	73.68	0.057	30.47	15.47
Mean (SD)	114.86 (22.4)	94.52 (19.9)	0.297 (0.39)	29.53 (2.9)	19.79 (8.4)
	- Year 2 -				
EF+N_{70}	23.66	176.43	0.216	54.21	20.62
HLF+M_{70}	36.33	158.54	0.574	54.22	18.45
LF	21.91	167.11	0.359	52.37	10.54
SP	20.28	163.7	0.573	31.66	12.18
Mean (SD)	25.55 (7.3)	166.45 (7.5)	0.431 (0.18)	48.12 (11.0)	15.45 (4.8)

5.7 Conclusions

There was no effect of FMS on spring wheat grain yield or N uptake, but differences from year to year across all systems were significant. Lower yields and N uptake in Year 1 could be attributed to increased precipitation that encouraged the downward movement of NO_3^- prior to cash crop planting together with wet, cool soil conditions that delayed root growth early in the growing season. Yearly differences in N uptake were reflected in the dynamics of surface soil

N_{min}, especially for FMS that were reliant on organic N sources alone. The EF+N_{70} FMS had the highest NO_3^- concentrations in drainage water through the fall and early winter, and again during the month of May. Emissions of N_2O were small across all systems in both years, although there was a response to manure addition in Year 1 during a period of sustained precipitation and the subsequent increase in soil water content. When compared to FMS that used only GMr alone, the use of supplemental N in the spring at a moderate rate (i.e., <70 kg avail. N ha^{-1}) did increase N_2O emissions in one year when applied as manure. Additionally, NO_3^- concentrations in drainage water increased when NH_4NO_3 fertilizer was applied with EF clover incorporation, but overall N losses were minimal from all FMS. Considering the additional energy required to produce and apply supplemental N in GMr systems, it is unclear if any additional benefits to crop yields would outweigh the environmental costs of applying additional N. In this two-year study with markedly different N supplying potential early in the growing season, delaying red clover incorporation in the fall (LF) proved to have a reliable N supplying ability and a low risk to in-season N_2O emissions and NO_3^- losses to drainage tiles. These results further highlight the difficulty in linking crop uptake with soil N supply when using legume GMr, but they do provide additional evidence of the benefit of GMr in supplying N to crops such as spring wheat or other N demanding crops in eastern Canada.

6. SYNTHESIS AND CONCLUSIONS

Soil processes and properties that control N dynamics in agroecosystems have received by far, the greatest amount of continued interest from soil scientists in the agricultural research community during the last century. The diversity of soil N processes that occur in the soil are known to be sensitive to soil properties, climatic conditions, and crop management practices (Zebarth et al., 2003), which makes reliable predictions of N availability from organic N sources extremely difficult. The lack of long-term studies that have focused on systems that use BNF as their main N input for soil fertility has left a gap in our understanding of the agricultural N cycle and the potential for N losses as N_2O (Basche et al., 2014; Skinner et al., 2014). Furthermore, systems that include a GMr phase in their rotation often involve large amounts of N mainly derived from BNF in the year before the cash crop. Therefore, off-season losses may be of greater interest, especially in areas that experience multiple freeze and thaw cycles.

A study with four unique fertility management systems (FMS) was conducted to examine how off-season environmental N losses in the GMr phase of the rotation varied by the timing and season of GMr incorporation together with the use of supplemental N. Off-season N losses were calculated as N_2O emissions from GMr and dissolved N_2O from sub-surface drainage tiles. Following the GMr phase in the rotation was spring wheat that allowed the GMr treatments to be evaluated with respect to grain yield and in-season N_2O emissions and full-year NO_3^- losses from the tile drainage system. Treatments (i.e., FMS) differed in management by the timing of GMr incorporation and the use of supplemental N as fertilizer and dairy manure in a system that removed aboveground biomass as forage. Because each phase of the rotation was present every year, we investigated the effects of GMr incorporation and the use of supplemental N in two distinct years that began in Apr. 2011 until June 2012 (Year 1), and again in Year 2 (June 2012–June 2013).

6.1 Summary of Findings

Nitrous oxide emissions were overall low from all treatments in this study if you were to compare these results to some intensively managed and higher yielding production systems on finer textured soils (e.g., Gregorich et al., 2005; Drury et al., 2014a), however were comparable to annual emissions on similar coarse textured soils (Gregorich et al., 2005; Burton et al., 2008). Overall, there was a 51% increase (Table 6.1) in N_2O emissions in Year 2 (1.37 ± 0.13 kg N ha^{-1}), as compared to Year 1 (0.91 ± 0.35 kg N ha^{-1}), but no difference in cumulative emissions among FMS (p>0.05) in either year of study. There was a wide range of off-season (Sep. to June) N_2O emissions among treatments and years from 0.26 and 1.22 kg N_2O-N ha^{-1} (Chapter 3). Overall, in-season N_2O emissions were low and ranged from 0.09 to 0.87 kg N_2O-N ha^{-1} over both years of study (Chapter 5). In-season N_2O emissions from the wheat crop did not differ among FMS, with the exception of an increase in N_2O on two sample dates in Year 1 following manure application in the HLF+M_{70} FMS. However, in-season emissions were only 33% of the annual emissions in both years of study (Fig. 6.1), with the greatest (p=0.11) emissions measured from the HLF+M_{70} system (0.87 kg N ha^{-1}) in Year 1, and from SP (p=0.02; 0.95 kg N ha^{-1}) in Year 2. Furthermore, the in-season sampling period was only 25% of the days used in annual emission calculations. In comparison, 37 and 43% of the off-season emissions were measured during the month of Mar. in Year 1 and 2, respectively, with SP having the highest emissions in Year 1 (0.44 kg N ha^{-1}) and the lowest in Year 2 (0.03 kg N ha^{-1}).

There were no differences in cumulative N_2O emissions among the three fall incorporated clover treatments in either year of study. However, the timing of N_2O emissions did differ across the three fall GMr incorporated treatments, such that, greater emission events occurred within a few weeks following clover incorporation. Spring GMr incorporation did increase cumulative N_2O emissions when compared to fall incorporation in the first year, yet decreased emissions in Year 2. We had a surprising pattern of N_2O emissions among treatments during spring thaw that could have been connected to soil freezing in Year 2, but freezing did not explain the pattern of N_2O emissions among treatments in Year 1. When freezing dynamics were similar among treatments in Year 1, the rate of thawing and the location of GMr residue likely had a greater influence on N_2O emissions from the SP treatment during spring thaw. A faster drainage rate during thawing in Year 2 suggests that hydraulic conductivity was impeded in Year 1. This result

is comparable to the contrasting results from studies that examine N_2O emissions from tilled and no-till systems at spring thaw (Lemke et al., 1999; Sheehy et al., 2013) and during the growing season (Ball et al., 1999; Grandy et al., 2006) that have reported differences between these systems were from an increase in WFPS and a decrease in gas diffusivity, and air-filled porosity (Rochette, 2008). It is quite possible that contrasting results among treatments in our study were influenced by soil hydraulic conductivity during spring behaving differently due to the amount of ice found within the soil profile, which would have altered the soil's air-filled porosity.

The amount of year to year variation in N_2O emissions is typical of most multi-year studies, but discussions regarding variable air or soil temperatures typically remain qualitative. In this study we summed the number of temperature degree hours above and below 0ºC for air and soil temperatures in an effort to quantify temporal differences in weather from year to year. The problem of relying on average daily or monthly temperature in year to year comparisons was most evident when describing weather during spring thaw. Mean air temperatures for Mar. (Table 3.4) were 0.4 and 0.2°C in Year 1 and 2, respectively, but there was actually 1034 more ΣFDH in the first two weeks of Year 1 and 806 more ΣTDH in the last two weeks of Year 1 when compared to Year 2. Daily variations in temperature during spring thaw can be substantial, and it is likely that these fluctuations will have an effect on soil processes associated with N_2O emissions.

Soil N_2O concentrations reached a maximum at the 15 and 30 cm depths in the SP and EF treatments on 8 Mar. 2012 with a mean (±SD) of 120±25 μL L^{-1}, while the highest concentrations at 50 cm were measured on 11 Mar. 2012 from the EF treatment (109 μL L^{-1}). In contrast, SP tillage had the lowest N_2O concentrations at all depths in Year 2, while all other treatments had a mean of 98±20 μL L^{-1} at 15 cm (6 Mar. 2013) and 113±37 μL L^{-1} (13 Mar. 2013) from the 30 to 50 cm depths. Accumulated N_2O from within the soil profile peaked before a significant drainage event during spring-thaw, which also increased dissolved N_2O content (g N ha^{-1}) from drainage tile water. These relationships at all soil depths suggests that soil processes throughout the profile were attributing to surface N_2O emissions, rather than being solely controlled by surface conditions. Furthermore, the dynamics of N_2O concentrations at soil depth proved to be a good indicator of potential N_2O production and emissions from the soil surface because the highest concentrations at depth also proved to be the FMS treatments with the greatest N_2O emissions (Chapters 4 and 5).

Table 6.1. Summary of N losses among fertility management systems (FMS) in Year 1 and 2: Surface N_2O emissions, N_2O dissolved in drainage water, and NO_3^- leaching.

Fertility Management Systems	Year 1						Year 2					
	N_2O Emissions						N_2O Emissions					
	Wheat (In-season)	GMr (Off-season)	Total	Dissolved N_2O	Leached NO_3^-	Total	Wheat (In-season)	GMr (Off-season)	Total	Dissolved N_2O	Leached NO_3^-	Total
	---------- kg N ha^{-1} ----------						---------- kg N ha^{-1} ----------					
Early fall + N_{70}	0.094	0.6 ab‡	0.694	0.033	32.05	32.77	0.156 b	1.221 a	1.377	0.057	20.62	24.14
Hayed late fall + M_{70}	0.872	0.475 b	1.347	0.04	17.94	19.32	0.437 ab	1.102 a	1.539	0.116	18.45	22.63
Late fall	0.164	0.414 b	0.578	0.021	13.69	14.32	0.336 ab	1.019 a	1.355	0.045	10.54	12.92
Spring	0.057	0.95 a	1.007	0.037	15.47	16.51	0.949 a	0.262 b	1.211	0.07	12.18	14.92
p-value	0.11	0.02	0.49	0.64	0.23	0.25	0.02	0.05	0.8	0.18	0.32	0.32

† Wheat in-season refers to N_2O measurements made from June to Sep. and off-season refer to measurements from Sep. to June.
‡ Means within columns with the same lower case letters are not significantly different at the $p \leq 0.05$ level of probability.

Increases in N_2O emissions were linked to CO_2 emissions, especially following clover incorporation in both years. However, high CO_2 emissions in the SP treatment in Year 1 were measured prior to spring incorporation when GMr residues and new biomass remained at the soil surface. There was also a positive relationship between soil CO_2 and N_2O concentrations at the 15- and 30-cm (r=0.7–0.76; p<0.001; n=46) depths, but the strength of this relationship diminished at 50 cm (r=0.16; p=0.26; n=46). Furthermore, daily surface emissions of N_2O had a positive relationship with the concentration of N_2O at the 30- (r=0.52; p<0.001; n=46) and 50-cm (r=0.45; p=0.001; n=46) soil depths throughout the winter. This relationship is substantial, as it shows that CO_2 production, as a result of increased microbial activity, contributed to the conditions needed for N_2O production due to limited diffusion of O_2 from above the soil surface.

To our knowledge, concentrations of dissolved N_2O measured from this study are within the range of the highest concentrations recorded in drainage water (Heincke and Kaupenjohann, 1999), and the first to make a connection of decreasing soil N_2O concentrations with increasing N_2O concentrations in drainage water. Overall, there was no difference in dissolved N_2O (µg N_2O-N L^{-1}) among treatments in Year 1 (p= 0.7) or Year 2 (p= 0.97), yet concentrations differed over time (p<0.001). Dissolved N_2O concentrations in drainage water increased during spring thaw as soil N_2O concentrations decreased with maximum values in drainage water of 167 and 269 µg N_2O-N L^{-1} on Mar. 9 and 13 of Year 1 and 2, respectively. The majority of the dissolved N_2O losses occurred during a major drainage event at the peak of the spring thaw, which also corresponded to a period with the highest daily surface flux measurements. Furthermore, total dissolved N_2O (g N_2O-N ha^{-1}) were similar to the maximum flux events (g N_2O-N ha^{-1} d^{-1}), which are known to be one of the greatest flux events of the year. Considering this, dissolved N_2O losses are a significant portion of spring thaw related N_2O losses in systems that have sub-surface drainage tiles.

There were no differences in wheat N uptake or grain yield among FMS in either year of study (Chapter 5), but overall there was a 76% increase (94.52 ± 19.9 to 166 ± 7.5 kg N ha^{-1}) in N uptake and a wheat grain yield that was 1.5 times greater in Year 2 (2.62 ± 0.27 Mg ha^{-1}), as compared to Year 1 (1.05 ± 0.12 Mg ha^{-1}). Lower yields and N uptake in Year 1 across all FMS could be attributed to spring soil conditions and increased precipitation that encouraged the downward movement of NO_3^- early in the spring. The soil content of N_{min} (kg N ha^{-1}) at the soil surface (0–15 cm) also reflected the difference in N availability between years, as there was

nearly twice the available soil N pool in Year 2 as compared to Year 1. However, the EF+N_{70} treatment had a greater N availability than all other FMS that were meeting N requirements through organic N sources in Year 1, indicating that GMr residue had already decomposed, and N_{min} had moved out of the root zone. These results further emphasize the necessity of continued long-term research in systems using differing FMS so that year to year variability due to weather can be properly recognized, rather than trying to extrapolate results from a year that was influenced by a unique, or uncharacteristic precipitation event. However, the relatively large N uptake values and grain yields in Year 2 that were comparable to systems using only inorganic fertilizers provides additional evidence of the potential of GMr to supply sufficient N to crops such as spring wheat, or other N demanding crops in eastern Canada.

Unfortunately, the portion of our study that was investigating drainage volumes from our field site did not start until July of 2011, which meant we were unable to quantify all of the leaching losses that likely occurred early in Year 1. However, the fact that very high N_{min} contents were measured below the root zone in the GMr phase of the rotation in May (Chapter 5), suggests that this N was unavailable for wheat uptake, and was likely lost to leaching. Furthermore, the trend of high N_{min} below the root zone in spring was evident in all phases of the rotation contained within the area drained by drainage tiles (data not shown), further evidence of the rapid movement of N_{min} below the root zone that was not just unique to the GMr phase.

The greatest NO_3^- losses (kg N ha^{-1}) to drainage water were measured following wheat harvest in response to high precipitation and limited plant uptake, but overall there were no difference among FMS in Year 1 (19.8 ± 8.4 kg N ha^{-1}), or Year 2 (15.4 ± 4.8 kg N ha^{-1}). However, the highest NO_3^- loads (kg N ha^{-1}) among FMS were measured in EF+N_{70} in Year 1 (32.1 kg N ha^{-1}) and Year 2 (20.6 kg N ha^{-1}), which was in agreement with the overall trend of EF+N_{70} having the highest NO_3^- concentrations from Oct. to Jan. and again in May (Chapter 5). The amount of RSMN after harvest is known to be a good indicator of potential off-season leaching losses, which supports our findings as well. The highest RSMN across all rotation phases were measured from EF+N_{70} plots (Table 5.6), while the phase with the soil profile N_{min} highest among all rotation phases were the 2nd year clover plots with 48.1 and 62.4 kg N ha^{-1} in Year 1 and 2, respectively.

In cropping systems that utilize a GMr phase in rotation with the cash crop, the season and timing of GMr incorporation, as well as the use of supplemental N, have a significant control

on the dynamics of soil N transformations and the balance of plant uptake with environmental losses. However, the greatest influence on these dynamics was from the variation in weather year to year. It is likely that improved drainage conditions via sub-surface drainage tiles contributed to the low N_2O emission from the surface, but also provided a pathway for substantial dissolved N_2O losses in drainage water during spring thaw.

6.2 Future Research

An area for future research would be the connection between GMr management and N losses to drainage water at the Brookside experimental site in Truro, NS (Appendix 1). This would complement the recent work by Drury (i.e., 2014a, and b) along with the steady stream of research from a group in Denmark (i.e., Chirinda et al., 2010b; Jabloun et al., 2015) who have been linking long-term cropping system management that involve legumes in rotation with their N losses as leached $NO_3^-$and N_2O emissions from the soil surface. Because of our experimental design, our evaluation of treatment effects on NO_3^- leaching and dissolved N_2O concentrations were limited to combined effects of varied GMr management within all phases of a 4-yr rotation, rather than being able to directly quantify the effect of the GMr tillage regimes alone on concentrations of N in drainage water. Additionally, the pattern of soil freezing with respect to the timing of tillage and the incorporation of GMr biomass (Chapter 3) and the accumulation of N_2O under a frozen basal layer (Chapter 4) could be further investigated in the other three phases of the rotation. Even with our estimates of dissolved N_2O content, which were suggested to be conservative, the amount of N_2O dissolved in drainage water that rapidly exited the drainage tiles was similar to a monthly N_2O emission total from the soil surface (Chapter 3), or a large single event at spring thaw (Risk et al. 2014). Therefore, a better understanding of the seasonal and field management influenced effects of increasing N_2O concentrations within the soil profile and the potential for losses as N_2O dissolved in drainage water in other phases of the rotation would help us understand this rarely investigated pathway for additional N_2O emissions.

The relationships between soil temperature and the magnitude of spring thaw N_2O emissions need further investigation. Because snow depth has such a strong control on soil freezing behavior it is difficult to relate the accumulation of soil freezing hours (ΣFDH; Chapter 3) to the magnitude of spring thaw N_2O emissions. The connection between substrate availability

and soil freezing on the disintegration of soil aggregates (Bullock et al., 1988; Dagesse et al., 2013), and microbial cell lysis or root turnover (Cleavitt et al., 2008; Elliott and Henry, 2009) with increased N_2O production does seem reasonable, but the effect of soil freezing on the hydrological controls of gas and water exchange deserves further attention.

Nitrous oxide production is intimately related to the availability of NO_3^- in the soil profile, yet the production and emission of N_2O from the soil surface and the soil content of NO_3^- are not necessarily connected. Alternatively, the availability of NO_3^- over time—as measured by the NO_3^- index (i.e., Burton et al. 2008; Snowdon et al., 2013)—is more of an integrative property that best describes N_2O emissions. A further examination of this relationship in these FMS would likely add further insight on the factors that control N_2O production. The idea of using a NO_3^- index became clear when looking at both the HLF and the LF treatment. The total and seasonal pattern of N_2O emissions was very similar among the HLF and LF treatments, although there was considerably less aboveground biomass returned to the HLF plots. When manure was applied to the HLF treatment in spring the similarities between these two treatments were not as clear when in-season N_2O emissions were considered. However, when off-season N_2O emissions are considered either the belowground contribution of substrate is more important for N_2O production, or the timing of tillage and the effects on soil pore size and distribution has a greater control. Further study on soil physical soil properties with respect to hydrological function would be warranted.

LITERATURE CITED

Abdollahi, L. and L.J. Munkholm. 2014. Tillage system and cover crop effects on soil quality: I. chemical, mechanical, and biological properties. Soil Sci. Soc. Am. J. 78:262–270.

Adviento-Borbe, M.A.A., J.P. Kaye, M.A. Bruns, M.D. McDaniel, M. McCoy and S. Harkcom. 2010. Soil greenhouse gas and ammonia emissions in long-term maize-based cropping systems. Soil Sci. Soc. Am. J. 74:1623–1634.

Ambus, P., U. Skiba, K. Butterbach-Bahl and M.A. Sutton. 2011. Reactive nitrogen and greenhouse gas flux interactions in terrestrial ecosystems. Plant Soil 343:1–3.

Amundson, R.G. and E.A. Davidson. 1990. Carbon-dioxide and nitrogenous gases in the soil atmosphere. J. Geochem. Explor. 38:13–41.

Andersen, M.K. and L.S. Jensen. 2001. Low soil temperature effects on short-term gross N mineralisation–immobilisation turnover after incorporation of a green manure. Soil Biol. Biochem. 33:511–521.

Arcand, M.M., R. Lemke, R.E. Farrell and J.D. Knight. 2014. Nitrogen supply from belowground residues of lentil and wheat to a subsequent wheat crop. Biol. Fertil. Soils 50:507–515.

Asgedom, H., M. Tenuta, D.N. Flaten, X. Gao, and E. Kebreab. 2014. Nitrous oxide emissions from a clay soil receiving granular urea formulations and dairy manure. Agron. J. 106:732–744.

Askegaard, M., J. Olesen and K. Kristensen. 2005. Nitrate leaching from organic arable crop rotations: Effects of location, manure and catch crop. Soil Use Manage. 21:181–188.

Askegaard, M., J.E. Olesen, I.A. Rasmussen and K. Kristensen. 2011. Nitrate leaching from organic arable crop rotations is mostly determined by autumn field management. Agric., Ecosyst. Environ. 142:149–160.

Aulakh, M.S., D.A. Rennie and E.A. Paul. 1983. The effect of various clover management practices on gaseous N losses and mineral N accumulation. Can. J. Soil Sci. 63:593–605.

Aulakh, M.S., J.W. Doran, D.T. Walters, A.R. Mosier and D.D. Francis. 1991. Crop residue type and placement effects on denitrification and mineralization. Soil Sci. Soc. Am. J. 55:1020–1025.

Baggs, E., R. Rees, K. Smith and A. Vinten. 2000. Nitrous oxide emission from soils after incorporating crop residues. Soil Use Manage. 16:82–87.

Balaine, N., T.J. Clough, M.H. Beare, S.M. Thomas, E.D. Meenken and J.G. Ross. 2013. Changes in relative gas diffusivity explain soil nitrous oxide flux dynamics. Soil Sci. Soc. Am. J. 77:1496–1505.

Ball, B.C., A. Scott and J.P. Parker. 1999. Field N_2O, CO_2 and CH_4 fluxes in relation to tillage, compaction and soil quality in Scotland. Soil Tillage Res. 53:29–39.

Ball, B.C., C.A. Watson and I. Crichton. 2007. Nitrous oxide emissions, cereal growth, N recovery and soil nitrogen status after ploughing organically managed grass/clover swards. Soil Use Manage. 23:145–155.

Ball, B.C., I. Crichton and G.W. Horgan. 2008. Dynamics of upward and downward N_2O and CO_2 fluxes in ploughed or no-tilled soils in relation to water-filled pore space, compaction and crop presence. Soil Tillage Res. 101:20–30.

Ball, B.C. 2013. Soil structure and greenhouse gas emissions: A synthesis of 20 years of experimentation. Eur. J. Soil Sci. 64:357–373.

Ball, B.C., B.S. Griffiths, C.F.E. Topp, R. Wheatley, R.L. Walker, R.M. Rees, C.A. Watson, H. Gordon, P.D. Hallett, B.M. McKenzie and I.M. Nevison. 2014. Seasonal nitrous oxide emissions from field soils under reduced tillage, compost application or organic farming. Agric. Ecosyst. Environ. 189:171–180.

Basche, A.D., F.E. Miguez, T.C. Kaspar and M.J. Castellano. 2014. Do cover crops increase or decrease nitrous oxide emissions? A meta-analysis. J. Soil Water Conserv. 69:471–482.

Baulch, H.M., S.L. Schiff, R. Maranger and P.J. Dillon. 2011. Nitrogen enrichment and the emission of nitrous oxide from streams. Global Biogeochem. Cycles 25. http://dx.doi.org/10.1029/2011GB004047

Beauchamp, E.G., J.T. Trevors and J.W. Paul. 1989. Carbon sources for bacterial denitrification. Adv. Soil Sci. 10:113–142.

Bedard-Haughn, A., L. Comeau and A. Sangster. 2013. Gross nitrogen mineralization in pulse-crop rotations on the Northern Great Plains. Nutr. Cycl. Agroecosys. 95:159–174.

Bélanger, G., J.R. Walsh, J.E. Richards, P.H. Milburn and N. Ziadi. 2000. Yield response of two potato cultivars to supplemental irrigation and N fertilization in New Brunswick. Am. J. Pot. Res. 77:11–21.

Belanger, G., N. Ziadi, J.R. Walsh, J.E. Richards and P.H. Milburn. 2003. Residual soil nitrate after potato harvest. J. Environ. Qual. 32:607–612.

Bergstrom, L. and H. Kirchmann. 2004. Leaching and crop uptake of nitrogen from nitrogen-15-labeled green manures and ammonium nitrate. J. Environ. Qual. 33:1786–1792.

Berry, P.M., E.A. Stockdale, R. Sylvester-Bradley, L. Philipps, K.A. Smith, E.I. Lord, C.A. Watson and S. Fortune. 2003. N, P and K budgets for crop rotations on nine organic farms in the UK. Soil Use Manage. 19:112–118.

Berry, P.M., R. Sylvester-Bradley, L. Philipps, D.J. Hatch, S.P. Cuttle, F.W. Rayns and P. Gosling. 2002. Is the productivity of organic farms restricted by the supply of available nitrogen? Soil Use Manage. 18:248–255.

Billings, S., D. Richter and J. Yarie. 1998. Soil carbon dioxide fluxes and profile concentrations in two boreal forests. Can. J. Forest Res. 28:1773–1783.

Blagodatsky, S. and P. Smith. 2012. Soil physics meets soil biology: Towards better mechanistic prediction of greenhouse gas emissions from soil. Soil Biol. Biochem. 47:78–92.

Blankinship, J.C. and S.C. Hart. 2012. Consequences of manipulated snow cover on soil gaseous emission and N retention in the growing season: A meta-analysis. Ecosphere 3. http://dx.doi.org/10.1890/ES11-00225.1

Bouwman, A., I. Fung, E. Matthews and J. John. 1993. Global analysis of the potential for N_2O production in natural soils. Global Biogeochem. Cycles 7:557–597.

Brady, N.C. and R.R. Weil. 2001. The nature and properties of soils. 13th ed. Prentice Hall, Upper Saddle River, NJ.

Breland, T. 1995. Green manuring with clover and ryegrass catch crops undersown in spring wheat: Effects on soil structure. Soil Use Manage. 11:163–167.

Brozyna, M.A., S.O. Petersen, N. Chirinda and J.E. Olesen. 2013. Effects of grass–clover management and cover crops on nitrogen cycling and nitrous oxide emissions in a stockless organic crop rotation. Agric., Ecosyst. Environ. 181:115–126.

Brumme, R., W. Borken and S. Finke. 1999. Hierarchical control on nitrous oxide emission in forest ecosystems. Global Biogeochem. Cycles 13:1137–1148.

Brummell, M.E., R.E. Farrell and S.D. Siciliano. 2012. Greenhouse gas soil production and surface fluxes at a high arctic polar oasis. Soil Biol. Biochem. 52:1–12.

Brummell, M.E., R.E. Farrell, S.P. Hardy and S.D. Siciliano. 2014. Greenhouse gas production and consumption in high arctic deserts. Soil Biol. Biochem. 68:158–165.

Bullock, M.S., S.D. Nelson and W.D. Kemper. 1988. Soil cohesion as affected by freezing, water content, time and tillage. Soil Sci. Soc. Am. J. 52:770–776.

Burton, D., D. Bergstrom, J. Covert, C. Wagner-Riddle and E. Beauchamp. 1997. Three methods to estimate N_2O fluxes as impacted by agricultural management. Can. J. Soil Sci. 77:125–134.

Burton, D.L. and E.G. Beauchamp. 1994. Profile nitrous oxide and carbon dioxide concentrations in a soil subject to freezing. Soil Sci. Soc. Am. J. 58:115–122.

Burton, D.L., B.J. Zebarth, K.M. Gillarn and J.A. MacLeod. 2008. Effect of split application of fertilizer nitrogen on N_2O emissions from potatoes. Can. J. Soil Sci. 88:229–239.

Burton, D.L., B.J. Zebarth, J.A. McLeod and C. Goyer. 2012. Nitrous oxide emissions from potato production and strategies to reduce them. p. 251–271. *In* Z. He, R. Larkin and W. Honeycutt (eds.) Sustainable potato production: Global case studies. Springer, New York, NY.

Butterbach-Bahl, K. and M. Dannenmann. 2011. Denitrification and associated soil N_2O emissions due to agricultural activities in a changing climate. Curr. Opin. Env. Sust. 3:389–395.

Butterbach-Bahl, K., E.M. Baggs, M. Dannenmann, R. Kiese and S. Zechmeister-Boltenstern. 2013. Nitrous oxide emissions from soils: How well do we understand the processes and their controls? Philos. T. Roy. Soc. B. 368. http://dx.doi.org/10.1098/rstb.2013.0122

Butterbach-Bahl, K., M. Kesik, P. Miehle, H. Papen and C. Li. 2004. Quantifying the regional source strength of N-trace gases across agricultural and forest ecosystems with process based models. Plant Soil 260:311–329.

Campbell, C.A., G.P. Lafond, A.J. VandenBygaart, R.P. Zentner, R. Lemke, W.E. May and C.B. Holzapfel. 2011. Effect of crop rotation, fertilizer and tillage management on spring wheat grain yield and N and P content in a thin black Chernozem: A long-term study. Can. J. Soil Sci. 91:467–483.

Carlsson, G. and K. Huss-Danell. 2003. Nitrogen fixation in perennial forage legumes in the field. Plant Soil 253:353–372.

Carter, M. 1996. Characterization of soil physical properties and organic matter under long-term primary tillage in a humid climate. Soil Till. Res. 38:251–263.

Carter, M.R. 1992. Characterizing the soil physical condition in reduced tillage systems for winter-wheat on a fine sandy loam using small cores. Can. J. Soil Sci. 72:395–402.

Carter, M.R. 1994. Influence of time of tillage on growth and yield parameters of spring cereals in a cool, humid climate. Soil Till. Res. 29:71–82.

Castellano, M.J., J.P. Schmidt, J.P. Kaye, C. Walker, C.B. Graham, H. Lin and C.J. Dell. 2010. Hydrological and biogeochemical controls on the timing and magnitude of nitrous oxide flux across an agricultural landscape. Global Change Biol. 16:2711–2720.

Chantigny, M., D. Prevost, D. Angers, R. Simard and F. Chalifour. 1998. Nitrous oxide production in soils cropped to corn with varying N fertilization. Can. J. Soil Sci. 78:589–596.

Chantigny, M.H., P. Rochette, D.A. Angers, S. Bittman, K. Buckley, D. Massé, G. Bélanger, N. Eriksen-Hamel and M. Gasser. 2010. Soil nitrous oxide emissions following band-incorporation of fertilizer nitrogen and swine manure. J. Environ. Qual. 39:1545–1553.

Cherkauer, K.A. and D.P. Lettenmaier. 1999. Hydrologic effects of frozen soils in the upper Mississippi River basin. J. Geophys. Res. 104:19599–19610.

Cho, C.M., D.L. Burton and C. Chang. 1997. Kinetic formulation of oxygen consumption and denitrification processes in soil. Can. J. Soil Sci. 77:253–260.

Chirinda, N., J. Olesen, J.R. Porter and P. Schjønning. 2010a. Soil properties, crop production and greenhouse gas emissions from organic and inorganic fertilizer-based arable cropping systems. Agric. Ecosyst. Environ. 139:584–594.

Chirinda, N., M.S. Carter, K.R. Albert, P. Ambus, J.E. Olesen, J.R. Porter and S.O. Petersen. 2010b. Emissions of nitrous oxide from arable organic and conventional cropping systems on two soil types. Agric. Ecosyst. Environ. 136:199–208.

Christensen, B.T. 2004. Tightening the nitrogen cycle. p. 47–67. *In* P. Schjønning, S. Elmholt and B.T. Christensen (eds.) Managing soil quality: Challenges in modern agriculture. CAB International, Wallingford.

Christensen, S. and J.M. Tiedje. 1990. Brief and vigorous N_2O production by soil at spring thaw. J. Soil Sci. 41:1–4.

Cicerone, R.J. 1989. Analysis of sources and sinks of atmospheric nitrous oxide (N_2O). J. Geophys. Res. 94:18265–18271.

Clark, K., M.H. Chantigny, D.A. Angers, P. Rochette and L. Parent. 2009. Nitrogen transformations in cold and frozen agricultural soils following organic amendments. Soil Biol. Biochem. 41:348–356.

Cleavitt, N.L., T.J. Fahey, P.M. Groffman, J.P. Hardy, K.S. Henry and C.T. Driscoll. 2008. Effects of soil freezing on fine roots in a northern hardwood forest. Can. J. Forest Res. 38:82–91.

Clough, T.J., R.R. Sherlock and D.E. Rolston. 2005. A review of the movement and fate of N_2O in the subsoil. Nutr. Cycling Agroecosyst. 72:3–11.

Conant, R.T., A.B. Berdanier and P.R. Grace. 2013. Patterns and trends in nitrogen use and nitrogen recovery efficiency in world agriculture. Global Biogeochem. Cycles 27:558–566.

Cookson, W.R., I.S. Cornforth and J.S. Rowarth. 2002. Winter soil temperature (2–15 °C) effects on nitrogen transformations in clover green manure amended or unamended soils; a laboratory and field study. Soil Biol. Biochem. 34:1401–1415.

Dagesse, D.F. 2013. Freezing cycle effects on water stability of soil aggregates. Can. J. Soil Sci. 93:473–483.

Dahlin, S., H. Kirchmann, T. Kätterer, S. Gunnarsson and L. Bergström. 2005. Possibilities for improving nitrogen use from organic materials in agricultural cropping systems. Ambio 34:288–295.

Davidson, E.A. and M.K. Firestone. 1988. Measurement of nitrous-oxide dissolved in soil solution. Soil Sci. Soc. Am. J. 52:1201–1203.

Davidson, E.A. 1991. Fluxes of nitrous oxide and nitric oxide from terrestrial ecosystems. p. 219–235. *In* J.E. Rogers and W.B. Whitman (eds.) Microbial production and consumption of greenhouse gases: methane, nitrogen oxides and halomethanes. American Society for Microbiology. Washington, DC.

Davidson, E.A., J.N. Galloway, N. Millar and A.M. Leach. 2014. N-related greenhouse gases in North America: Innovations for a sustainable future. Curr. Opin. Environ. Sustainability 9–10:1–8.

Dawson, J.C., D.R. Huggins and S.S. Jones. 2008. Characterizing nitrogen use efficiency in natural and agricultural ecosystems to improve the performance of cereal crops in low-input and organic agricultural systems. Field Crops Res. 107:89–101.

De Jong, R., J.Y. Yang, C.F. Drury, E.C. Huffman, V. Kirkwood and X.M. Yang. 2007. The indicator of risk of water contamination by nitrate-nitrogen. Can. J. Soil Sci. 87:179–188.

De Jong, R., C. F. Drury, J. Y. Yang, and C. A. Campbell. 2009. Risk of water contamination by nitrogen in Canada as estimated by the IROWC-N model. J. Environ. Manage. 90:3169–3181

de Vries, F.T. and R.D. Bardgett. 2012. Plant–microbial linkages and ecosystem nitrogen retention: Lessons for sustainable agriculture. Front. Ecol. Environ. 10:425–432.

Dessureault-Rompre, J., B.J. Zebarth, D.L. Burton, A. Georgallas, M. Sharifi, G.A. Porter, G. Moreau, Y. Leclerc, W.J. Arsenault, T.L. Chow and C.A. Grant. 2012. Prediction of soil nitrogen supply in potato fields using soil temperature and water content information. Soil Sci. Soc. Am. J. 76:936–949.

Dessureault-Rompre, J., B. Zebarth, D.L. Burton and A. Georgallas. 2015. Predicting soil nitrogen supply from soil properties. Can. J. Soil Sci. (in press).

Dietzel, R., D. Wolfe and J.E. Thies. 2011. The influence of winter soil cover on spring nitrous oxide emissions from an agricultural soil. Soil Biol. Biochem. 43:1989–1991.

Dinnes, D., D. Karlen, D. Jaynes, T. Kaspar, J. Hatfield, T. Colvin and C. Cambardella. 2002. Nitrogen management strategies to reduce nitrate leaching in tile-drained midwestern soils. Agron. J. 94:153–171.

Doran, J.W., D.G. Fraser, M.N. Culik and W.C. Liebhardt. 1987. Influence of alternative and conventional agricultural management on soil microbial processes and nitrogen availability. Am. J. Alter. Agric. 2:99–106.

Dowdell, R.J., J.R. Burford and R. Crees. 1979. Losses of nitrous-oxide dissolved in drainage water from agricultural land. Nature 278:342–343.

Drinkwater, L.E., P. Wagoner and M. Sarrantonio. 1998. Legume-based cropping systems have reduced carbon and nitrogen losses. Nature 396:262–265.

Drury, C.F., J.Y. Yang, R. De Jong, X.M. Yang, E.C. Huffman, V. Kirkwood and K. Reid. 2007. Residual soil nitrogen indicator for agricultural land in Canada. Can. J. Soil Sci. 87:167–177.

Drury, C.F., X.M. Yang, W.D. Reynolds and N.B. McLaughlin. 2008. Nitrous oxide and carbon dioxide emissions from monoculture and rotational cropping of corn, soybean and winter wheat. Can. J. Soil Sci. 88:163–174.

Drury, C.F., W.D. Reynolds, C.S. Tan, N.B. McLaughlin, X.M. Yang, W. Calder, T.O. Oloya and J.Y. Yang. 2014a. Impact of 49-51 years of fertilization and crop rotation on growing season nitrous oxide emissions, nitrogen uptake and corn yields. Can. J. Soil Sci. 94:421–433.

Drury, C.F., C.S. Tan, T.W. Welacky, W.D. Reynolds, T.Q. Zhang, T.O. Oloya, N.B. McLaughlin and J.D. Gaynor. 2014b. Reducing nitrate loss in tile drainage water with cover crops and water-table management systems. J. Environ. Qual. 43:587–598.

Duxbury, J.M., D.R. Bouldin, R.E. Terry and R.L. Tate. 1982. Emissions of nitrous oxide from soils. Nature 298:462–464.

Ellert, B.H. and H.H. Janzen. 2008. Nitrous oxide, carbon dioxide and methane emissions from irrigated cropping systems as influenced by legumes, manure and fertilizer. Can. J. Soil Sci. 88:207–217.

Elliott, A.C. and H.A.L. Henry. 2009. Freeze-thaw cycle amplitude and freezing rate effects on extractable nitrogen in a temperate old field soil. Biol. Fertility Soils 45:469–476.

Eriksen, J., M. Askegaard and K. Kristensen. 2004. Nitrate leaching from an organic dairy crop rotation: The effects of manure type, nitrogen input and improved crop rotation. Soil Use Manage. 20:48–54.

Evans, J., G. Scott, D. Lemerle, A. Kaiser, B. Orchard, G. Murray and E. Armstrong. 2003. Impact of legume 'break' crops on the yield and grain quality of wheat and relationship with soil mineral N and crop N content. Aust. J. Agric. Res. 54:777–788.

Fang, C. and J. Moncrieff. 1999. A model for soil CO_2 production and transport 1: Model development. Agric. for. Meteorol. 95:225–236.

Farrell, R.E., E. De Jong and J.A. Elliott. 2002. Gas sampling and analysis. p. 1076–1111. *In* J.H. Dane and C.G. Topp (eds.) Methods of soil analysis: Part 4 physical methods. Soil Science Society of America, Madison, WI.

Firestone, M.K., M.S. Smith, R.B. Firestone and J.M. Tiedje. 1979. Influence of nitrate, nitrite, and oxygen on the composition of the gaseous products of denitrification in soil. Soil Sci. Soc. Am. J. 43:1140–1144.

Firestone, M.K. and E.A. Davidson. 1989. Microbiological basis of NO and N_2O production and consumption in soil. p. 7–21. *In* M.O. Andreae and D. Schimel (eds.) Exchange of trace gases between terrestrial ecosystems and the atmosphere. John Wiley & Sons, Chichester, UK.

Francis, G., R. Haynes, G. Sparling, D. Ross and P. Williams. 1992. Nitrogen mineralization, nitrate leaching and crop growth following cultivation of a temporary leguminous pasture in autumn and winter. Fert. Res. 33:59–70.

Francis, G.S., R.J. Haynes and P.H. Williams. 1995. Effects of the timing of plowing-in temporary leguminous pastures and 2 winter cover crops on nitrogen mineralization, nitrate leaching and spring wheat growth. J. Agric. Sci. 124:1–9.

Franzluebbers, A. 2002. Water infiltration and soil structure related to organic matter and its stratification with depth. Soil Tillage Res. 66:197–205.

Froseth, R.B., A.K. Bakken, M.A. Bleken, H. Riley, R. Pommeresche, K. Thorup-Kristensen and S. Hansen. 2014. Effects of green manure herbage management and its digestate from biogas production on barley yield, N recovery, soil structure and earthworm populations. Eur. J. Agron. 52:90–102.

Furon, A.C., C. Wagner-Riddle, C.R. Smith and J.S. Warland. 2008. Wavelet analysis of wintertime and spring thaw CO_2 and N_2O fluxes from agricultural fields. Agric. for. Meteorol. 148:1305–1317.

Galloway, J.N., E.B. Cowling, S.P. Seitzinger and R.H. Socolow. 2002. Reactive nitrogen: Too much of a good thing? Ambio. 31:60–63.

Gao, X., N. Rajendran, M. Tenuta, A. Dunmola and D.L. Burton. 2014. Greenhouse gas accumulation in the soil profile is not always related to surface emissions in a prairie pothole agricultural landscape. Soil Sci. Soc. Am. J. 78:805–817.

Gentry, L.E., S.S. Snapp, R.F. Price and L.F. Gentry. 2013. Apparent red clover nitrogen credit to corn: Evaluating cover crop introduction. Agron. J. 105:1658–1664.

Gillam, K.M., B.J. Zebarth and D.L. Burton. 2008. Nitrous oxide emissions from denitrification and the partitioning of gaseous losses as affected by nitrate and carbon addition and soil aeration. Can. J. Soil Sci. 88:133–143.

Glenn, A.J., M. Tenuta, B.D. Amiro, S.E. Maas and C. Wagner-Riddle. 2012. Nitrous oxide emissions from an annual crop rotation on poorly drained soil on the Canadian prairies. Agric. for. Meteorol. 166–167:41–49.

Goldberg, S.D., K. Knorr and G. Gebauer. 2008. N_2O concentration and isotope signature along profiles provide deeper insight into the fate of N_2O in soils. Isotopes Environ. Health Stud. 44:377–391.

Goldberg, S., W. Borken and G. Gebauer. 2010. N_2O emission in a Norway spruce forest due to soil frost: Concentration and isotope profiles shed a new light on an old story. Biogeochemistry 97:21–30.

Goodroad, L.L. and D.R. Keeney. 1984. Nitrous-oxide emissions from soils during thawing. Can. J. Soil Sci. 64:187–194.

Grandy, A.S., T.D. Loecke, S. Parr and G.P. Robertson. 2006. Long-term trends in nitrous oxide emissions, soil nitrogen, and crop yields of till and no-till cropping systems. J. Environ. Qual. 35:1487–1495.

Grant, R.F., R.C. Izaurralde and D.S. Chanasyk. 1990. Soil temperature under conventional and minimum tillage: simulation and experimental verification. Can. J. Soil Sci. 70:289–304.

Gregorich, E., P. Rochette, A. VandenBygaart and D. Angers. 2005. Greenhouse gas contributions of agricultural soils and potential mitigation practices in eastern Canada. Soil Tillage Res. 83:53–72.

Gregorich, E.G., P. Rochette, P. St-Georges, U.F. McKim and C. Chan. 2008. Tillage effects on N_2O emission from soils under corn and soybeans in eastern Canada. Can. J. Soil Sci. 88:153–161.

Griffin, T.S. 2008. Nitrogen availability. p. 613–646. *In* J. Schepers and W.R. Raun (eds.) Nitrogen in agricultural systems. Agron. Monogr. 49. ASA, CSSA, SSSA, Madison, WI.

Groffman, P., C. Driscoll, T. Fahey, J. Hardy, R. Fitzhugh and G. Tierney. 2001. Colder soils in a warmer world: A snow manipulation study in a northern hardwood forest ecosystem. Biogeochemistry 56:135–150.

Hamdi, S., F. Moyano, S. Sall, M. Bernoux and T. Chevallier. 2013. Synthesis analysis of the temperature sensitivity of soil respiration from laboratory studies in relation to incubation methods and soil conditions. Soil Biol. Biochem. 58:115–126.

Harrysson Drotz, S., E.L. Tilston, T. Sparrman, J. Schleucher, M. Nilsson and M.G. Öquist. 2009. Contributions of matric and osmotic potentials to the unfrozen water content of frozen soils. Geoderma 148:392–398.

Hartmann, D.L., A.M.G. Klein Tank, M. Rusticucci, L.V. Alexander, S. Brönnimann, Y. Charabi, F.J. Dentener, E.J. Dlugokencky, D.R. Easterling, A. Kaplan, B.J. Soden, P.W. Thorne, M. Wild and P.M. Zhai. 2013. Observations: Atmosphere and surface. p. 159–254. *In* T.F. Stocker, D. Qin, G.-K. Plattner, M. Tignor, S.K. Allen, J. Boschung, A. Nauels, Y. Xia, V. Bex and P.M. Midgley (eds.) Climate change 2013: The physical science basis. contribution of working group I to the fifth assessment report of the intergovernmental panel on climate change. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Hatch, D., A. Joynes, S. Roderick, M. Shepherd and G. Goodlass. 2014. Effects of cutting, mulching and applications of farmyard manure on the supply of nitrogen from a red clover/grass sward. Org. Agri. 4:15–24.

Heincke, M. and M. Kaupenjohann. 1999. Effects of soil solution on the dynamics of N_2O emissions: A review. Nutr. Cycling Agroecosyst. 55:133–157.

Henry, H.L. 2008. Climate change and soil freezing dynamics: Historical trends and projected changes. Clim. Change. 87:421–434.

Hepperly, P., R. Seidel, D. Pimentel, J. Hanson and D.J. Douds. 2007. Organic farming enhances soil carbon and its benefits. p. 129–153. *In* J. Kimble, C. Rice, D. Reed, S. Mooney, R. Follet and R. Lal (eds.). Soil carbon management: Economic, environmental, and societal benefits. CRC Press, Boca Raton, FL.

Hoyle, F.C., D.V. Murphy and I.R.P. Fillery. 2006. Temperature and stubble management influence microbial CO_2–C evolution and gross N transformation rates. Soil Biol. Biochem. 38:71–80.

Huang, Y., J. Zou, X. Zheng, Y. Wang and X. Xu. 2004. Nitrous oxide emissions as influenced by amendment of plant residues with different C:N ratios. Soil Biol. Biochem. 36:973–981.

Hutchinson, G.L. and G.P. Livingston. 1993. Use of chamber systems to measure trace gas fluxes. p. 63–78. *In* D.E. Rolston, J.M. Duxbury, L.A. Harper and A.R. Mosier (eds.) Agricultural ecosystem effects on trace gases and global climate change. American Society of Agronomy. Madison. WI.

Ireson, A.M., d.K. van, G. Ferguson, U. Nachshon and H.S. Wheater. 2013. Hydrogeological processes in seasonally frozen northern latitudes: Understanding, gaps and challenges. Hydrogeol. J. 21:53–66.

Iwata, Y., M. Nemoto, S. Hasegawa, Y. Yanai, K. Kuwao and T. Hirota. 2011. Influence of rain, air temperature, and snow cover on subsequent spring-snowmelt infiltration into thin frozen soil layer in northern Japan. J. Hydrol. 401:165–176.

Jabloun, M., K. Schelde, F. Tao and J.E. Olesen. 2015. Effect of temperature and precipitation on nitrate leaching from organic cereal cropping systems in Denmark. Eur. J. Agron. 62:55–64.

Jacinthe, P. A. and W.A. Dick. 1996. Use of silicone tubing to sample nitrous oxide in the soil atmosphere. Soil Biol. Biochem. 28:721–726.

Jacinthe, P., W. Dick and L. Owens. 2002. Overwinter soil denitrification activity and mineral nitrogen pools as affected by management practices. Biol. Fertil. Soils 36:1–9.

Jahangir, M.M.R., P. Johnston, M.I. Khalil, J. Grant, C. Somers and K.G. Richards. 2012. Evaluation of headspace equilibration methods for quantifying greenhouse gases in groundwater. J. Environ. Manage. 111:208–212.

Janzen, H.H. 1998. The health of our air. Research Branch, Agriculture and Agri-Food Canada, Ontario.

Jensen, E.S., M.B. Peoples, R.M. Boddey, P.M. Gresshoff, H. Hauggaard-Nielsen, B.J.R. Alves and M.J. Morrison. 2012. Legumes for mitigation of climate change and the provision of feedstock for biofuels and biorefineries. A review. Agro. Sust. Dev. 32:329–364.

Jiang, Y., B. Zebarth and J. Love. 2011. Long-term simulations of nitrate leaching from potato production systems in Prince Edward Island, Canada. Nutr. Cycling Agroecosyst. 91:307–325.

Johansson, E., A. Hussain, R. Kuktaite, S.C. Andersson and M.E. Olsson. 2014. Contribution of organically grown crops to human health. Int. J. Environ. Res. Publ. Health 11:3870–3893.

Johnson, J.M.F., D. Archer and N. Barbour. 2010. Greenhouse gas emission from contrasting management scenarios in the northern corn belt. Soil Sci. Soc. Am. J. 74:396–406.

Kaiser, E., K. Kohrs, M. Kucke, E. Schnug, J. Munch and O. Heinemeyer. 1998. Nitrous oxide release from arable soil: Importance of perennial forage crops. Biol. Fertility Soils 28:36–43.

Kane, D.L., K.M. Hinkel, D.J. Goering, L.D. Hinzman and S.I. Outcalt. 2001. Non-conductive heat transfer associated with frozen soils. Global Planet. Change 29:275–292.

Kariyapperuma, K.A., C. Wagner-Riddle, A.C. Furon and C. Li. 2011. Assessing spring thaw nitrous oxide fluxes simulated by the DNDC model for agricultural soils. Soil Sci. Soc. Am. J. 75:678–690.

Katayanagi, N. and R. Hatano. 2012. N_2O emissions during the freezing and thawing periods from six fields in a livestock farm, southern Hokkaido, Japan. Soil Sci. Plant Nutr. 58:261–271.

Kaut, A.H.E.E., H.E. Mason, A. Navabi, J.T. O'Donovan and D. Spaner. 2009. Performance and stability of performance of spring wheat variety mixtures in organic and conventional management systems in western Canada. J. Agric. Sci. 147:141–153.

Kay, B. and A. VandenBygaart. 2002. Conservation tillage and depth stratification of porosity and soil organic matter. Soil Till. Res. 66:107–118.

Keeney, D.R. and D.W. Nelson. 1982. Nitrogen-inorganic forms. p. 643–698. *In* A.L. Page, R.H. Miller and D.R. Keeney (eds.) Methods of soil analysis. part 2 – chemical and microbiological properties. 2nd ed. Agronomy, Madison, WI.

Knight, J.D., R. Buhler, J.Y. Leeson and S.J. Shirtliffe. 2010. Classification and fertility status of organically managed fields across Saskatchewan, Canada. Can. J. Soil Sci. 90:667–678.

Kojima, Y., J.L. Heitman, G.N. Flerchinger and R. Horton. 2013. Numerical evaluation of a sensible heat balance method to determine rates of soil freezing and thawing. Vadose Zone J. 12. http://dx.doi.org/10.2136/vzj2012.0053

Kong, A.Y., S. Fonte, C. Kessel and J. Six. 2007. Soil aggregates control N cycling efficiency in long-term conventional and alternative cropping systems. Nutr. Cycl. Agroecos. 79:45–58.

Kool, D.M., J. Dolfing, N. Wrage and J.W. Van Groenigen. 2011. Nitrifier denitrification as a distinct and significant source of nitrous oxide from soil. Soil Biol. Biochem. 43:174–178.

Koponen, H.T., T. Jaakkola, M.M. Keinanen-Toivola, S. Kaipainen, J. Tuomainen, K. Servomaa and P.J. Martikainen. 2006. Microbial communities, biomass, and activities in soils as affected by freeze thaw cycles. Soil Biol. Biochem. 38:1861–1871.

Korsaeth, A., T.M. Henriksen and L.R. Bakken. 2002. Temporal changes in mineralization and immobilization of N during degradation of plant material: Implications for the plant N supply and nitrogen losses. Soil Biol. Biochem. 34:789–799.

Korsaeth, A. 2008. Relations between nitrogen leaching and food productivity in organic and conventional cropping systems in a long-term field study. Agric. Ecosyst. Environ. 127:177–188.

Kramer, S., J. Reganold, J. Glover, B. Bohannan and H. Mooney. 2006. Reduced nitrate leaching and enhanced denitrifier activity and efficiency in organically fertilized soils. Proc. Natl. Acad. Sci. U. S. A. 103:4522–4527.

Lahti, T. and P.J. Kuikman. 2003. The effect of delaying autumn incorporation of green manure crop on N mineralization and spring wheat (Triticum aestivum L.) performance. Nutr. Cycl. Agroecosys. 65:265–280.

Larionova, A.A., I.V. Yevdokimov and S.S. Bykhovets. 2007. Temperature response of soil respiration is dependent on concentration of readily decomposable C. Biogeosciences 4:1073–1081.

Lavoie, M., L. Kellman and D. Risk. 2013. The effects of clear-cutting on soil CO_2, CH_4, and N_2O flux, storage and concentration in two Atlantic temperate forests in Nova Scotia, Canada. For. Ecol. Manage. 304:355–369.

Lemke, R.L., R.C. Izaurralde, M. Nyborg and E.D. Solberg. 1999. Tillage and N source influence soil-emitted nitrous oxide in the Alberta parkland region. Can. J. Soil Sci. 79:15–24.

Liebman, M., R.L. Graef, D. Nettleton and C.A. Cambardella. 2012. Use of legume green manures as nitrogen sources for corn production. Renew. Agr. Food Syst. 27:180–191.

Ling, F. and T. Zhang. 2004. A numerical model for surface energy balance and thermal regime of the active layer and permafrost containing unfrozen water. Cold Reg. Sci. Technol. 38:1–15.

Linn, D.M. and J.W. Doran. 1984. Effect of water-filled pore-space on carbon-dioxide and nitrous-oxide production in tilled and nontilled soils. Soil Sci. Soc. Am. J. 48:1267–1272.

Liu, K., A.M. Hammermeister, M.H. Entz, T. Astatkie, P.R. Warman and R.C. Martin. 2010. Nitrogen availability in an organic potato crop following 3-year transition under contrasting farming systems. J. Sustainable Agric. 34:82–835.

Loes, A., T.M. Henriksen, R. Eltun and H. Sjursen. 2011. Repeated use of green-manure catch crops in organic cereal production - grain yields and nitrogen supply. Acta Agric Scand B. 61:164–175.

Lynch, D. 2009. Environmental impacts of organic agriculture: A Canadian perspective. Can. J. Plant Sci. 89:621–628.

Lynch, D.H. 2014. Sustaining soil organic carbon, soil quality and soil health in organic field crop management systems. p. 107–132. *In* R.C. Martin and R. MacRae (eds.) Managing energy, nutrients, and pests in organic field crops. CRC Press, Boca Raton, FL.

Lynch, D.H., M. Sharifi, A. Hammermeister, and D. Burton. 2012. Nitrogen management in organic potato production. p. 209–231. *In* H. Zhongi, R.P. Larkin and C.W. Honeycutt (eds.) Sustainable potato production: Global case studies. Springer Science + Business Media, New York.

Lynch, D.H., R. MacRae and R.C. Martin. 2011. The carbon and global warming potential impacts of organic farming: Does it have a significant role in an energy constrained world? Sustainability 3:322–362.

Lynch, D.H., Z. Zheng, B.J. Zebarth and R.C. Martin. 2008. Organic amendment effects on tuber yield, plant N uptake and soil mineral N under organic potato production. Renew. Agr. Food Syst. 23:250–259.

Lynch, D.H., J. Sumner and R.C. Martin. 2014. Framing the social, ecological and economic goods and services derived from organic agriculture in the Canadian context. p. 347–365. *In* S. Bellon and S. Penvern (eds.) Organic farming, prototype for sustainable agriculture. Springer, Netherlands.

Lynch, J. 1980. Effects of organic-acids on the germination of seeds and growth of seedlings. Plant Cell Environ. 3:255–259.

Lund, V. 2006. Natural living - a precondition for animal welfare in organic farming. Livestock Science 100:71–83.

Ma, W.K., A. Bedard-Haughn, S.D. Siciliano and R.E. Farrell. 2008. Relationship between nitrifier and denitrifier community composition and abundance in predicting nitrous oxide emissions from ephemeral wetland soils. Soil Biol. Biochem. 40:1114–1123.

Maas, S.E., A.J. Glenn, M. Tenuta and B.D. Amiro. 2013. Net CO_2 and N_2O exchange during perennial forage establishment in an annual crop rotation in the Red River Valley, Manitoba. Can. J. Soil Sci. 93:639–652.

Magid, J., O. Henriksen, K. Thorup-Kristensen and T. Mueller. 2001. Disproportionately high N-mineralisation rates from green manures at low temperatures – implications for modeling and management in cool temperate agro-ecosystems. Plant Soil 228:73–82.

Magkos, F., F. Arvaniti and A. Zampelas. 2003. Organic food: Nutritious food or food for thought? A review of the evidence. Int. J. Food Sci. Nutr. 54:357–371.

Maharjan, B. and R.T. Venterea. 2013. Nitrite intensity explains N management effects on N_2O emissions in maize. Soil Biol. Biochem. 66:229–238.

Maier, M. and H. Schack-Kirchner. 2014. Using the gradient method to determine soil gas flux: A review. Agric. for. Meteorol. 192–193:78–95.

Makoto, K., T. Kajimoto, L. Koyama, G. Kudo, H. Shibata, Y. Yanai and J.H.C. Cornelissen. 2013. Winter climate change in plant–soil systems: Summary of recent findings and future perspectives. Ecol. Res. 29:593–606.

Malhi, S.S. and M. Nyborg. 1986. Increase in mineral N in soils during winter and loss of mineral N during early spring in north-central Alberta. Can. J. Soil Sci. 66:397–409.

Maljanen, M., A.-R. Kohonen, P. Virkajarvi and P.J. Martikainen. 2007. Fluxes and production of N_2O, CO_2 and CH_4 in boreal agricultural soil during winter as affected by snow cover. Tellus B 59:853–859.

Matzner, E. and W. Borken. 2008. Do freeze–thaw events enhance C and N losses from soils of different ecosystems? A review. Eur. J. Soil Sci. 59:274–284.

McDaniel, M.D., L.K. Tiemann and A.S. Grandy. 2014a. Does agricultural crop diversity enhance soil microbial biomass and organic matter dynamics? A meta-analysis. Ecol. Appl. 24:560–570.

McDaniel, M.D., A.S. Grandy, L.K. Tiemann and M.N. Weintraub. 2014b. Crop rotation complexity regulates the decomposition of high and low quality residues. Soil Biol. Biochem. 78:243–254.

Milburn, P., J. MacLeod and B. Sanderson. 1997. Control of fall nitrate leaching from early harvested potatoes on Prince Edward Island. Can. Agric. Eng. 39:263–271.

Miller, M.N., B.J. Zebarth, C.E. Dandie, D.L. Burton, C. Goyer and J.T. Trevors. 2008. Crop residue influence on denitrification, N_2O emissions and denitrifier community abundance in soil. Soil Biol. Biochem. 40:2553–2562.

Miller, M.N., B.J. Zebarth, C.E. Dandie, D.L. Burton, C. Goyer and J.T. Trevors. 2009. Influence of liquid manure on soil denitrifier abundance, denitrification, and nitrous oxide emissions. Soil Sci. Soc. Am. J. 73:760–768.

Minamikawa, K., S. Eguchi, S. Nishimura, H. Ihara, M. Maeda, K. Yagi and M. Komada. 2013. Groundwater-induced emissions of nitrous oxide through the soil surface and from subsurface drainage in an andosol upland field: A monolith lysimeter study. Soil Sci. Plant Nutr. 59:87–95.

Mitchell, D.C., M.J. Castellano, J.E. Sawyer and J. Pantoja. 2013. Cover crop effects on nitrous oxide emissions: Role of mineralizable carbon. Soil Sci. Soc. Am. J. 77: 1765–1773.

Mkhabela, M.S., R. Gordon, D. Burton, A. Madani and W. Hart. 2008. Nitrous oxide emissions and soil mineral nitrogen status following application of hog slurry and inorganic fertilisers to acidic soils under forage grass. Can. J. Soil Sci. 88:145–151.

Morkved, P.T., P. Dorsch, T.M. Henriksen and L.R. Bakken. 2006. N_2O emissions and product ratios of nitrification and denitrification as affected by freezing and thawing. Soil Biol. Biochem. 38:3411–3420.

Mosier, A., C. Kroeze, C. Nevison, O. Oenema, S. Seitzinger and O. Van Cleemput. 1998. Closing the global N_2O budget: Nitrous oxide emissions through the agricultural nitrogen cycle: OECD/IPCC/IEA phase II development of IPCC guidelines for national greenhouse gas inventory methodology. Nutr. Cycling Agroecosyst. 52:225–248.

Mutegi, J.K., L.J. Munkholm, B.M. Petersen, E.M. Hansen and S.O. Petersen. 2010. Nitrous oxide emissions and controls as influenced by tillage and crop residue management strategy. Soil Biol. Biochem. 42:1701–1711.

Myhre, G., D. Shindell, F.-M. Bréon, W. Collins, J. Fuglestvedt, J. Huang, D. Koch, J.-F. Lamarque, D. Lee, B. Mendoza, T. Nakajima, A. Robock, G. Stephens, T. Takemura and H. Zhang. 2013. Anthropogenic and natural radiative forcing. p. 659–740. *In* T.F. Stocker, D. Qin, G.-K. Plattner, M. Tignor, S.K. Allen, J. Boschung, A. Nauels, Y. Xia, V. Bex and P.M. Midgley (eds.) Climate change 2013: The physical science basis. contribution of working group I to the fifth assessment report of the intergovernmental panel on climate change. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Nass, H., L. Shugar and M. Etienne. 2001. AC Helena spring wheat. Can. J. Plant Sci. 81:277–279.

Nelson, K.L., D.H. Lynch and G. Boiteau. 2009. Assessment of changes in soil health throughout organic potato rotation sequences. Agric. Ecosyst. Environ. 131:220–228.

Németh, D.D., C. Wagner-Riddle and K.E. Dunfield. 2014. Abundance and gene expression in nitrifier and denitrifier communities associated with a field scale spring thaw N_2O flux event. Soil Biol. Biochem. 73:1–9.

Nevison, C. 2000. Review of the IPCC methodology for estimating nitrous oxide emissions associated with agricultural leaching and runoff. Chemos. Global Change Sci. 2:493–500.

Novak, S.M. and J.L. Fiorelli. 2010. Greenhouse gases and ammonia emissions from organic mixed crop-dairy systems: A critical review of mitigation options. Agron. Sustain. Dev. 30:215–236.

Olesen, J., K. Schelde, A. Weiske, M. Weisbjerg, W. Asman and J. Djurhuus. 2006. Modelling greenhouse gas emissions from European conventional and organic dairy farms. Agric. Ecosyst. Environ. 112:207–220.

Osterholz, W.R., C.J. Kucharik, J.L. Hedtcke and J.L. Posner. 2014. Seasonal nitrous oxide and methane fluxes from grain- and forage-based production systems in Wisconsin, USA. J. Environ. Qual. 43:1833–1843.

Outcalt, S.I., F.E. Nelson and K.M. Hinkel. 1990. The zero-curtain effect: Heat and mass transfer across an isothermal region in freezing soil. Water Resour. Res. 26:1509–1516.

Parkin, G., A.P. von Bertoldi and A.J. McCoy. 2013. Effect of tillage on soil water content and temperature under Freeze–Thaw conditions. Vadose Zone J. 12: http://dx.doi.org/10.2136/vzj2012.0075.

Parkin, T.B. 2008. Effect of sampling frequency on estimates of cumulative nitrous oxide emissions. J. Environ. Qual. 37:1390–1395.

Pelster, D.E., M.H. Chantigny, P. Rochette, D.A. Angers, C. Rieux and A. Vanasse. 2012. Nitrous oxide emissions respond differently to mineral and organic nitrogen sources in contrasting soil types. J. Environ. Qual. 41:427–435.

Pelster, D.E., M.H. Chantigny, P. Rochette, D.A. Angers, J. Laganière, B. Zebarth and C. Goyer. 2013. Crop residue incorporation alters soil nitrous oxide emissions during freeze–thaw cycles. Can. J. Soil Sci. 93:1–11.

Pennock, D.J., T. T Yates and J.T. Braidek. 2006. Towards optimum sampling for regional-scale N_2O emission monitoring in Canada. Can. J. Soil Sci. 86:441–450.

Peoples, M.B., D.F. Herridge and J.K. Ladha. 1995. Biological nitrogen fixation: An efficient source of nitrogen for sustainable agricultural production? Plant Soil 174:3–28.

Peoples, M. and J. Baldock. 2001. Nitrogen dynamics of pastures: Nitrogen fixation inputs, the impact of legumes on soil nitrogen fertility, and the contributions of fixed nitrogen to Australian farming systems. Aust. J. Exp. Agric. 41:327–346.

Peoples, M.B., J. Brockwell, D.F. Herridge, I.J. Rochester, B.J.R. Alves, S. Urquiaga, R.M. Boddey, F.D. Dakora, S. Bhattarai, S.L. Maskey, C. Sampet, B. Rerkasem, D.F. Khan, H. Hauggaard-Nielsen and E.S. Jensen. 2009. The contributions of nitrogen-fixing crop legumes to the productivity of agricultural systems. Symbiosis 48:1–17.

Petersen, S.O., P. Ambus, L. Elsgaard, P. Schjønning and J.E. Olesen. 2013. Long-term effects of cropping system on N_2O emission potential. Soil Biol. Biochem. 57:706–712.

Phillips, R. 2008. Denitrification in cropping systems at sub-zero soil temperatures. A review. Agron. Sustain. Dev. 28:87–93.

Phillips, R.L. 2007. Organic agriculture and nitrous oxide emissions at sub-zero soil temperatures. J. Environ. Qual. 36:23–30.

Rasouli, S., J.K. Whalen and C.A. Madramootoo. 2014. Review: Reducing residual soil nitrogen losses from agroecosystems for surface water protection in Quebec and Ontario, Canada: Best management practices, policies and perspectives. Can. J. Soil Sci. 94:109–127.

Ravishankara, A.R., J.S. Daniel and R.W. Portmann. 2009. Nitrous oxide (N_2O): The dominant ozone-depleting substance emitted in the 21st century. Science 326:123–125.

Reay, D.S., A.C. Edwards and K.A. Smith. 2004a. Determinants of nitrous oxide emission from agricultural drainage waters. Water Air Soil Pol. 4:107–115.

Reay, D.S., K.A. Smith and A.C. Edwards. 2004b. Nitrous oxide in agricultural drainage waters following field fertilisation. Water Air Soil Pol. 4:437–451.

Rees, R.M., J.A. Baddeley, A. Bhogal, B.C. Ball, D.R. Chadwick, M. Macleod, A. Lilly, V.A. Pappa, R.E. Thorman, C.A. Watson and J.R. Williams. 2013. Nitrous oxide mitigation in UK agriculture. Soil Sci. Plant Nutr. 59:3–15.

Refsgaard, K., N. Halberg and E.S. Kristensen. 1998. Energy utilization in crop and dairy production in organic and conventional livestock production systems. Agri. Sys. 57:599–630.

Risk, D., L. Kellman and H. Beltrami. 2002. Carbon dioxide in soil profiles: Production and temperature dependence. Geophys. Res. Lett. 29(6).1087. http://dx.doi.org/10.1029/2001GL014002

Risk, N., C. Wagner-Riddle, A. Furon, J. Warland and C. Blodau. 2014. Comparison of simultaneous soil profile N_2O concentration and surface N_2O flux measurements overwinter and at spring thaw in an agricultural soil. Soil Sci. Soc. Am. J. 78:180–193.

Risk, N., D. Snider and C. Wagner-Riddle. 2013. Mechanisms leading to enhanced soil nitrous oxide fluxes induced by freeze–thaw cycles. Can. J. Soil Sci. 93:101–114.

Roberts, C.J., D.H. Lynch, R.P. Voroney, R.C. Martin and S.D. Juurlink. 2008. Nutrient budgets of Ontario organic dairy farms. Can. J. Soil Sci. 88:107–113.

Robertson, G.P. and P.M. Vitousek. 2009. Nitrogen in agriculture: Balancing the cost of an essential resource. Annual Review of Environment and Resources 34:97–125.

Rochette, P. and H. Janzen. 2005. Towards a revised coefficient for estimating N_2O emissions from legumes. Nutr. Cycling Agroecosyst. 73:171–179.

Rochette, P., D. Angers, M. Chantigny, N. Bertrand and D. Cote. 2004. Carbon dioxide and nitrous oxide emissions following fall and spring applications of pig slurry to an agricultural soil. Soil Sci. Soc. Am. J. 68:1410–1420.

Rochette, P. 2008. No-till only increases N_2O emissions in poorly-aerated soils. Soil Tillage Res. 101:97–100.

Rochette, P., D.A. Angers, M.H. Chantigny, B. Gagnon and N. Bertrand. 2008. N_2O fluxes in soils of contrasting textures fertilized with liquid and solid dairy cattle manures. Can. J. Soil Sci. 88:175–187.

Roper, J.D. 2009. Influence of tillage practices on surface and subsurface nitrogen emissions from agricultural soils. MSc. Thesis. Dalhousie University, Canada.

Roper, J.D., D.L. Burton, A. Madani and G.W. Stratton. 2013. A simple method for quantifying dissolved nitrous oxide in tile drainage water. Can. J. Soil Sci. 93:59–64.

Rover, M., O. Heinemeyer and E. Kaiser. 1998. Microbial induced nitrous oxide emissions from an arable soil during winter. Soil Biol. Biochem. 30:1859–1865.

Russow, R., C.F. Stange and H.-U. Neue. 2009. Role of nitrite and nitric oxide in the processes of nitrification and denitrification in soil: Results from ^{15}N tracer experiments. Soil Biol. Biochem. 41:785–795.

Sanchez, J., T. Willson, K. Kizilkaya, E. Parker and R. Harwood. 2001. Enhancing the mineralizable nitrogen pool through substrate diversity in long term cropping systems. Soil Sci. Soc. Am. J. 65:1442–1447.

Sanderson, J. and J. MacLeod. 1994. Soil nitrate profile and response of potatoes to fertilizer-N in relation to time of incorporation of lupin (*Lupinus-albus*). Can. J. Soil Sci. 74:241–246.

Sanderson, J., J. MacLeod and J. Kimpinski. 1999. Glyphosate application and timing of tillage of red clover affects potato response to N, soil N profile, and root and soil nematodes. Can. J. Soil Sci. 79:65–72.

Savard, M.M., D. Paradis, G. Somers, S. Liao and E. Van Bochove. 2007. Winter nitrification contributes to excess NO_3^- in groundwater of an agricultural region: A dual-isotope study. Water Resour. Res. 43: W06422. http://dx.doi.org/10.1029/2006WR005469

Schimel, J.P. and J. Bennett. 2004. Nitrogen mineralization: Challenges of a changing paradigm. Ecology. 85: 591–602.

Schmidt, C.S., D.J. Richardson and E.M. Baggs. 2011. Constraining the conditions conducive to dissimilatory nitrate reduction to ammonium in temperate arable soils. Soil Biol. Biochem. 43:1607–1611.

Sharifi, M., B.J. Zebarth, D.L. Burton, C.A. Grant and G.A. Porter. 2008. Organic amendment history and crop rotation effects on soil nitrogen mineralization potential and soil nitrogen supply in a potato cropping system. Agron. J. 100:1562–1572.

Sharifi, M., B.J. Zebarth, G.A. Porter, D.L. Burton and C.A. Grant. 2009. Soil mineralizable nitrogen and soil nitrogen supply under two-year potato rotations. Plant Soil 320:267–279.

Sheehy, J., J. Six, L. Alakukku and K. Regina. 2013. Fluxes of nitrous oxide in tilled and no-tilled boreal arable soils. Agric. Ecosyst. Environ. 164:190–199.

Shirtliffe, S.J. and D. Benaragama. 2014. Sometimes you need a big hammer: Evaluating and appraising selected non-herbicidal weed control methods in an integrated weed management system. p. 149–168. *In* R.C. Martin and R. MacRae (eds.) Managing energy, nutrients, and pests in organic field crops. CRC Press, Boca Raton, FL.

Six, J., E.T. Elliott and K. Paustian. 2000. Soil macroaggregate turnover and microaggregate formation: A mechanism for C sequestration under no-tillage agriculture. Soil Biol. Biochem. 32:2099–2103.

Skinner, C., A. Gattinger, A. Muller, P. Mäder, A. Flieβbach, M. Stolze, R. Ruser and U. Niggli. 2014. Greenhouse gas fluxes from agricultural soils under organic and non-organic management — A global meta-analysis. Sci. Total Environ. 468–469:553–563.

Smith, J., C. Wagner-Riddle and K. Dunfield. 2010. Season and management related changes in the diversity of nitrifying and denitrifying bacteria over winter and spring. Appl. Soil Ecol. 44:138–146.

Smith, K.A., T. Ball, F. Conen, K.E. Dobbie, J. Massheder and A. Rey. 2003. Exchange of greenhouse gases between soil and atmosphere: Interactions of soil physical factors and biological processes. Eur. J. Soil Sci. 54:779–791.

Smith, W., B. Grant, R. Desjardins, R. Lemke and C. Li. 2004. Estimates of the interannual variations of N_2O emissions from agricultural soils in Canada. Nutr. Cycling Agroecosyst. 68:37–45.

Smith, P., D. Martino, Z. Cai, D. Gwary, H. Janzen, P. Kumar, B. McCarl, S. Ogle, F. O'Mara, C. Rice, B. Scholes, O. Sirotenko, M. Howden, T. McAllister, G. Pan, V. Romanenkov, U. Schneider, S. Towprayoon, M. Wattenbach and J. Smith. 2008. Greenhouse gas mitigation in agriculture. Philos. Trans. R. Soc. Lond. B Biol. Sci. 363:789–813.

Snowdon, E., B.J. Zebarth, D.L. Burton, C. Goyer and P. Rochette. 2013. Growing season N_2O emissions from two-year potato rotations in a humid environment in New Brunswick, Canada. Can. J. Soil Sci. 93:279–294.

St. Luce, M., J.K. Whalen, N. Ziadi and B.J. Zebarth. 2011. Nitrogen dynamics and indices to predict soil nitrogen supply in humid temperate soils. Adv. Agron. 112:55–102. http://dx.doi.org/10.1016/B978-0-12-385538-1.00002-0

Stopes, C., E. Lord, L. Philipps and L. Woodward. 2002. Nitrate leaching from organic farms and conventional farms following best practice. Soil Use Manage. 18:256–263.

Strudley, M.W., T.R. Green and J.C. Ascough II. 2008. Tillage effects on soil hydraulic properties in space and time: State of the science. Soil Tillage Res. 99:4–48.

Syvasalo, E., K. Regina, M. Pihlatie and M. Esala. 2004. Emissions of nitrous oxide from boreal agricultural clay and loamy sand soils. Nutr. Cycling Agroecosyst. 69:155–165.

Tatti, E., C. Goyer, M. Chantigny, S. Wertz, B.J. Zebarth, D.L. Burton and M. Filion. 2014. Influences of over winter conditions on denitrification and nitrous oxide-producing microorganism abundance and structure in an agricultural soil amended with different nitrogen sources. Agric. Ecosyst. Environ. 183:47–59.

Thilakarathna, R.M.M.S., Y.A. Papadopoulos, A.V. Rodd, A.N. Gunawardena, S.A.E. Fillmore and B. Prithiviraj. 2012. Characterizing nitrogen transfer from red clover populations to companion bluegrass under field conditions. Can. J. Plant Sci. 92:1163–1173.

Thorup-Kristensen, K., J. Magid and L. Jensen. 2003. Catch crops and green manures as biological tools in nitrogen management in temperate zones. Advances in Agronomy, Vol 79:227–302.

Thorup-Kristensen, K. 2006. Effect of deep and shallow root systems on the dynamics of soil inorganic N during 3-year crop rotations. Plant Soil. 288:233–248.

Thorup-Kristensen, K. and D.B. Dresboll. 2010. Incorporation time of nitrogen catch crops influences the N effect for the succeeding crop. Soil Use Manage. 26:27–35.

Tisdall, J.M. and J.M. Oades. 1982. Organic-matter and water-stable aggregates in soils. J. Soil Sci. 33:141–163.

van Bochove, E., D. Prevost and F. Pelletier. 2000a. Effects of freeze–thaw and soil structure on nitrous oxide produced in a clay soil. Soil Sci. Soc. Am. J. 64:1638–1643.

van Bochove, E., H. Jones, N. Bertrand and D. Prevost. 2000b. Winter fluxes of greenhouse gases from snow-covered agricultural soil: Intra-annual and interannual variations. Global Biogeochem. Cycles. 14:113–125.

van Bochove, E., G. Theriault, P. Rochette, H. Jones and J. Pomeroy. 2001. Thick ice layers in snow and frozen soil affecting gas emissions from agricultural soils during winter. J. Geophys. Res. Atmos. 106:23061–23071.

van Kessel, C., R. Venterea, J. Six, M.A. Adviento-Borbe, B. Linquist and K.J. van Groenigen. 2013. Climate, duration, and N placement determine N_2O emissions in reduced tillage systems: A meta-analysis. Global Change Biol. 19:33–44.

van der Weerden, T.J., F.M. Kelliher and C.A.M. de Klein. 2012. Influence of pore size distribution and soil water content on nitrous oxide emissions. Soil Res. 50:125–135.

van Groenigen, J.W., G.L. Velthof, O. Oenema, K.J. Van Groenigen and C. Van Kessel. 2010. Towards an agronomic assessment of N_2O emissions: A case study for arable crops. Eur. J. Soil Sci. 61:903–913.

Vankoughnett, M. and H.L. Henry. 2013. Combined effects of soil freezing and N addition on losses and interception of N over winter and summer. Ecosystems 16:694–703.

Venterea, R.T. and D.E. Rolston. 2000. Nitric and nitrous oxide emissions following fertilizer application to agricultural soil: Biotic and abiotic mechanisms and kinetics. Journal of Geophysical Research-Atmospheres 105:15117–15129.

Venterea, R.T. 2007. Nitrite-driven nitrous oxide production under aerobic soil conditions: Kinetics and biochemical controls. Global Change Biol. 13:1798–1809.

Voroney, R.P. 2007. The soil habitat. p. 25–49. *In* E.A. Paul (ed.) Soil microbiology, ecology and biochemistry (third edition). Third Edition ed. Academic Press. San Diego, CA.

Wagner-Riddle, C. and G. Thurtell. 1998. Nitrous oxide emissions from agricultural fields during winter and spring thaw as affected by management practices. Nutr. Cycling Agroecosyst. 52:151–163.

Wagner-Riddle, C., A. Furon, N.L. Mclaughlin, I. Lee, J. Barbeau, S. Jayasundara, G. Parkin, P. von Bertoldi and J. Warland. 2007. Intensive measurement of nitrous oxide emissions from a corn–soybean–wheat rotation under two contrasting management systems over 5 years. Global Change Biol. 13:1722–1736.

Wagner-Riddle, C., Q.C. Hu, E. van Bochove and S. Jayasundara. 2008. Linking nitrous oxide flux during spring thaw to nitrate denitrification in the soil profile. Soil Sci. Soc. Am. J. 72:908–916.

Wagner-Riddle, C., J. Rapai, J. Warland and A. Furon. 2010. Nitrous oxide fluxes related to soil freeze and thaw periods identified using heat pulse probes. Can. J. Soil Sci. 90:409–418.

Walley, F.L., G.W. Clayton, P.R. Miller, P.M. Carr and G.P. Lafond. 2007. Nitrogen economy of pulse crop production in the Northern Great Plains. Agron. J. 99:1710–1718.

Watanabe, K., T. Kito, S. Dun, J.Q. Wu, R.C. Greer and M. Flury. 2013. Water infiltration into a frozen soil with simultaneous melting of the frozen layer. Vadose Zone J. 12: http://dx.doi.org/10.2136/vzj2011.0188

Watson, C.A., D. Atkinson, P. Gosling, L.R. Jackson and F.W. Rayns. 2002. Managing soil fertility in organic farming systems. Soil Use Manage. 18:239–247.

Webb, K.T., R.L. Thompson, G.J. Beke and J.L. Nowland. 1991. Soils of Colchester County, Nova Scotia. Report no. 19. Nova Scotia Soil Survey. Research Branch, Agriculture Canada, Ottawa, Ontario, Canada.

Weiss, R.F. and B.A. Price. 1980. Nitrous oxide solubility in water and seawater. Mar. Chem. 8:347–359.

Wertz, S., C. Goyer, B.J. Zebarth, D.L. Burton, E. Tatti, M.H. Chantigny and M. Filion. 2013. Effects of temperatures near the freezing point on N_2O emissions, denitrification and on the abundance and structure of nitrifying and denitrifying soil communities. FEMS Microbiol. Ecol. 83:242–254.

Whalen, J.K. and L. Sampedro. 2010. Soil ecology and management. CAB International, UK.

Williams, M., D. Helmig and P. Blanken. 2009. White on green: Under-snow microbial processes and trace gas fluxes through snow, Niwot Ridge, Colorado Front Range. Biogeochemistry. 95:1–12.

Woodley, A., Y. Audette, T. Fraser, M. Arcand, P. Voroney, J.D. Knight and D.H. Lynch. 2014. Nitrogen and phosphorus fertility management in organic field crop production. p. 149–168. *In* R.C. Martin and R. MacRae (eds.) Managing energy, nutrients, and pests in organic field crops. CRC Press, Boca Raton, FL.

Wrage, N., G.L. Velthof, M.L. van Beusichem and O. Oenema. 2001. Role of nitrifier denitrification in the production of nitrous oxide. Soil Biol. Biochem. 33:1723-1732.

Wuebbles, D.J. 2009. Nitrous oxide: No laughing matter. Science 326:56–57.

Wunderlin, P., J. Mohn, A. Joss, L. Emmenegger and H. Siegrist. 2012. Mechanisms of N_2O production in biological wastewater treatment under nitrifying and denitrifying conditions. Water Res. 46:1027–1037.

Yanai, Y., T. Hirota, Y. Iwata, M. Nemoto, O. Nagata and N. Koga. 2011. Accumulation of nitrous oxide and depletion of oxygen in seasonally frozen soils in northern japan – snow cover manipulation experiments. Soil Biol. Biochem. 43:1779–1786.

Zebarth, B., Y. Leclerc, G. Moreau, R. Gareau and P. Milburn. 2003. Soil inorganic nitrogen content in commercial potato fields in New Brunswick. Can. J. Soil Sci. 83:425–429.

Zebarth, B.J., P. Rochette, and D.L. Burton. 2008. N_2O emissions from spring barley production as influenced by fertilizer nitrogen rate. Can. J. Soil Sci. 88:197–205.

Zebarth, B.J., C.F. Drury, N. Tremblay and A.N. Cambouris. 2009. Opportunities for improved fertilizer nitrogen management in production of arable crops in eastern Canada: A review. Can. J. Soil Sci. 89:113–132.

Zikeli, S., S. Gruber, C. Teufel, K. Hartung and W. Claupein. 2013. Effects of reduced tillage on crop yield, plant available nutrients and soil organic matter in a 12-year long-term trial under organic management. Sustainability. 5:3876–3894.

APPENDIX 1. SCHEMATIC OF THE BROOKSIDE EXPERIMENTAL SITE FOR ORGANIC AGRICULTURE AT DALHOUSIE UNIVERSITY'S AGRICULTURAL CAMPUS, TRURO, NOVA SCOTIA, CANADA.

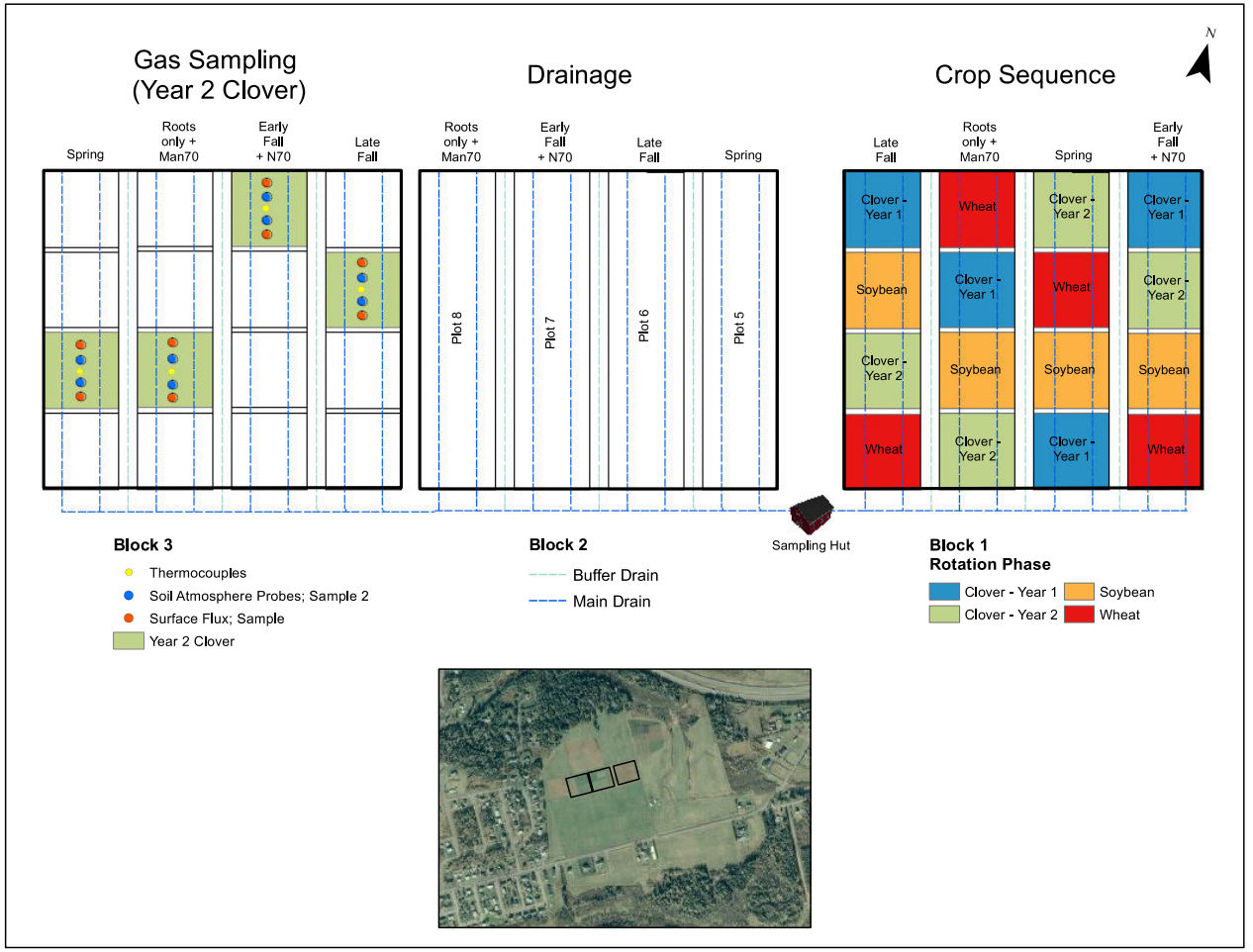

Fig. 1. Block 1 is an example of the rotation phase with four sub-plots (16 x 16 m) in each main plot (n=12; 16 x 75 m). Spacing of sub-surface drainage tiles in each main plot is shown in Block 2 with tile lines running into the water-sampling hut. Gas sampling locations are presented in Block 3 with the location for soil temperature measurements labeled as "Thermocouple".